United States
Environmental Protection
Agency

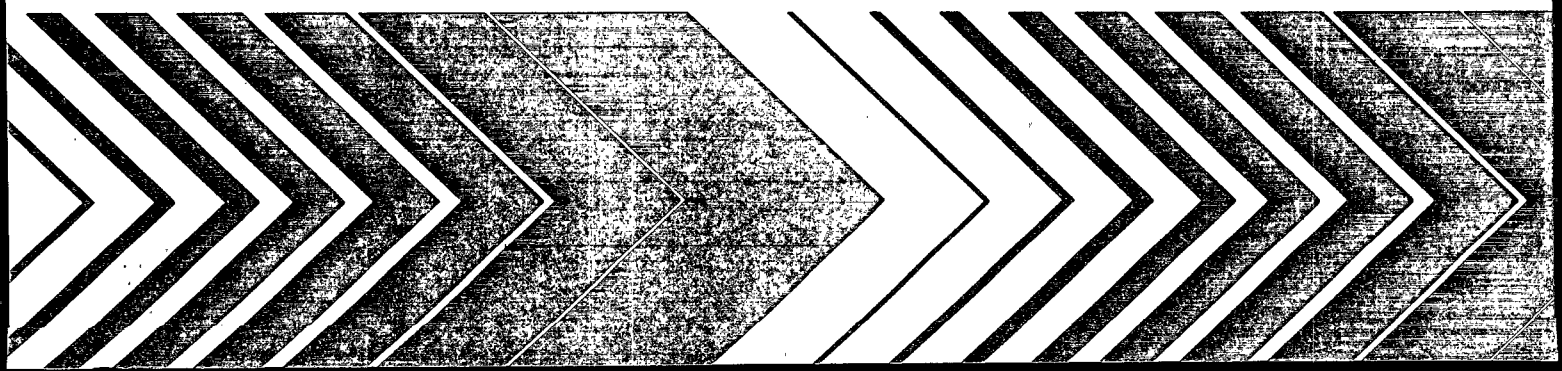
Office of Health and
Environmental Assessment
Washington DC 20460

EPA/600/6-88/006
May 1986

Research and Development

EPA

Development of a Qualitative Pathogen Risk Assessment Methodology for Municipal Sludge Landfilling



EPA/600/6-88/006
May 1986

DEVELOPMENT OF A QUALITATIVE PATHOGEN RISK ASSESSMENT METHODOLOGY FOR MUNICIPAL SLUDGE LANDFILLING

Environmental Criteria and Assessment Office
Office of Health and Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Cincinnati, OH 45268

DISCLAIMER

This document has been reviewed in accordance with the U.S. Environmental Protection Agency policy and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

PREFACE

Municipal wastewater sludges contain a wide variety of bacteria, viruses, protozoa, helminths and fungi. There is a need to develop a risk assessment methodology to investigate the potential risks from microbiological pathogens present in municipal sludge disposed of in landfills. Survival characteristics of pathogens are critical factors in assessing the risks associated with potential transport of microorganisms from the sludge-soil matrix to the groundwater environment of landfills. Various models are discussed for predicting microbial die-off.

DOCUMENT DEVELOPMENT

Larry Fradkin, Document Manager
Environmental Criteria and Assessment Office
Office of Health and Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Cincinnati, OH 45268

Dr. Charles Gerba, Author
Department of Microbiology and Immunology
University of Arizona
Tucson, AZ 85721

Dr. Pat V. Scarpino, Co-Document Manager
Civil and Environmental Engineering Department
University of Cincinnati
Cincinnati, OH 45268

Scientific Reviewer(s)

Dr. Richard S. Engelbrecht
Department of Civil Engineering
University of Illinois at Urbana-Champaign
Urbana, IL 61801

Dr. Elmer Akin
Dr. Walt Jakubowski
Dr. Norm Kowal
Health Effects Research Laboratory
Office of Health Research
Office of Research and Development
U.S. Environmental Protection Agency
Cincinnati, OH 45268

Dr. Carl A. Brunner
Dr. Joe Farrell
Dr. Albert D. Venosa
Wastewater Research Division
Water Engineering Research Laboratory
Office of Environmental Engineering and Technology Demonstration
Office of Research and Development
U.S. Environmental Protection Agency
Cincinnati, OH

Dr. Mark D. Sobsey
Department of Environmental Sciences and Engineering
School of Public Health
University of North Carolina
Chapel Hill, NC 27514

Technical Editor

Judith A. Olsen
Environmental Criteria and Assessment Office
U.S. Environmental Protection Agency
Cincinnati, OH 45268

TABLE OF CONTENTS

TABLE OF CONTENTS (cont.)

LIST OF TABLES

LIST OF TABLES (cont.)

LIST OF TABLES (cont.)

LIST OF FIGURES

LIST OF ABBREVIATIONS AND SYMBOLS

HAV	Hepatitis a virus
MID	Minimum infectious dose
MPN	Most probable number
PFU	Plague-forming units
sp.	Species (singular)
spp.	Species (plural)
t_{90}	Time for 90% inactivation
$TCID_{50}$	Dose at which 50% of inoculated tissue cultures are infected

1. INTRODUCTION

Municipal sludge landfilling is defined for purposes of this assessment as the burying of sludge, that is, the application of sludge to the land and subsequent internment by applying a layer of cover soil over sludge. To be defined as a landfill, the thickness of the soil cover must be greater than the depth of the plow zone (U.S. EPA, 1978). For this reason, subsurface injection of sludge is a landspreading, not a landfilling, operation and is not considered in this document. Municipal sludges are often co-disposed with municipal solid wastes from household and industrial sources. This document only considers landfills solely used for municipal sludge disposal.

Municipal sludges are known to contain microorganisms capable of causing serious illness and mortality in humans (Evans, 1982). Many of these organisms are responsible for waterborne disease in the United States today (Craun, 1986). The number of documented cases of waterborne disease in the United States has been on the increase for almost two decades (Craun, 1986); contaminated groundwater is responsible for almost half of the outbreaks each year (Keswick and Gerba, 1980). Enteric pathogens are the most common cause of food and waterborne illness in the United States today (NRC, 1985; Craun, 1986). Between 1971 and 1980 chemicals were responsible for <5% of all reported waterborne illnesses in the United States (Craun, 1986). Thus, it is essential that methodologies be developed to assess risks from microorganisms during sludge disposal so that appropriate criteria can be established for their management. This document addresses risks from microbiological pathogens and the development of methodologies by which sludge disposal criteria may be derived to minimize these risks.

2. SLUDGE CHARACTERISTICS AND LANDFILLING METHODS

2.1. SLUDGE CHARACTERISTICS

Municipal sludge is a complex mixture of solids of biological and mineral origin that are removed from wastewater in sewage treatment plants. Sludge is a by-product of physical (primary treatment), biological (activated sludge, trickling filters) and physiochemical (precipitation with lime, ferric chloride or alum) treatment of wastewater. Many of the pathogenic microorganisms present in raw wastewaters find their way into municipal sludges. Treatment of these sludges by anaerobic or aerobic digestion and/or dewatering reduces the number of pathogens, but significant numbers remain. The type of treatment determines the concentration of pathogens and the relative risk of disposal.

Only dewatered sludges with solids contents $\geq$15% are considered suitable for disposal in sludge-only landfills. Sludges having solids contents <15% usually will not support cover material. In some operations soil may be added as a bulking agent to a low solids sludge to produce a sludge suitable for disposal at sludge-only landfills. Sludges may be dewatered by a number of processes including drying beds, vacuum filtration, pressure filtration, centrifugation and heat drying. Chemicals such as alum, lime, ferric chloride or synthetic polyelectrolytes are added to improve the dewatering characteristics of the sludge. In general, only stabilized sludges are recommended for landfilling, but this is not required in all states (U.S. EPA, 1978).

Stabilization of sludges may be accomplished by either aerobic or anaerobic digestion, lime addition, heat, wet oxidation or incineration.

2.2. SLUDGE LANDFILLING METHODS

Several different methods of disposal are used at sludge-only landfills. These are listed in Table 2-1. The type of method utilized is dependent on the characteristics of the sludge and the nature of the site. Recommended site and sludge conditions are shown in Table 2-2. The different methods of disposal may have different risks associated with them, and a description of each is included.

2.2.1. Trench. For trenches, subsurface excavation is required so that the sludge can be placed entirely below the original ground surface. Soil is used only for cover and is not used as a sludge bulking agent. The sludge is usually dumped directly into the trench from hauling vehicles, and cover is generally applied over sludge the same day that it is received. The soil excavated during trench construction in most cases is sufficient for cover applications. Application is either to narrow or wide trenches. Narrow trenches are defined as having widths <10 ft (3.0 m); wide trenches are defined as having widths >10 ft (3.0 m) (U.S. EPA, 1978). Trench depth is a function of depth to groundwater and bedrock, sidewall stability and equipment limitations.

For narrow trenches it is recommended that sludge solids content be at least 15-20% for 2-3 ft (0.6-0.9 m) widths and 20-28% for 3-10 ft (0.9-3.0 m) widths (U.S. EPA, 1978). However, a review of landfills in operation indicates that sludges with a solids content of as low as 3% are disposed of by this method (U.S. EPA, 1978). The main advantage of a narrow trench is its ability to handle sludge with a relatively low solids content. Generally, application rates for narrow trenches are less than for other methods. It is impractical to install liners in narrow trench operations.

TABLE 2-1

Sludge Landfilling Methods*

Trench Only	Area Fill
Narrow trench	Area fill mound
Wide trench	Area fill layer
	Diked containment

*Source: U.S. EPA, 1978

TABLE 2-2

Sludge and Site Conditions Recommended for Landfilling*

Method	Sludge Solids Content	Sludge Characteristics	Hydrogeology	Ground Slope
Narrow trench	15-28%	unstabilized or stabilized	deep groundwater and bedrock	<20%
Wide trench	>20%	unstabilized or stabilized	deep groundwater and bedrock	<10%
Area fill mound	>20%	stabilized	shallow groundwater or bedrock	Suitable for steep terrain as long as level area is prepared for mounding
Area fill layer	>15%	unstabilized or stabilized	shallow groundwater or bedrock	Suitable for medium slopes but level ground preferred
Diked containment	>20%	stabilized	shallow groundwater or bedrock	Suitable for steep terrain as long as a level area is prepared inside dikes

*Source: U.S. EPA, 1978

In wide-trench application, sludges with a solids content of ≥20% are recommended. Wide-trench methods are less land intense than narrow-trench methods and liners can be installed to contain sludge moisture and protect groundwater.

Both stabilized and unstabilized sludges can be disposed of by trench application.

2.2.2. Area Fill. For area fills, sludge is placed above the original ground surface. Because excavation is not required and sludge is not placed below the surface, area fill applications are often used in areas with shallow groundwater or bedrock. The solids content of sludge as received is not necessarily limited. However, because the sidewall containment (available in a trench) is lacking and equipment must be supported atop the sludge in most area fills, sludge stability and bearing capacity must be relatively good. To achieve these qualities, soil is mixed with the sludge as a bulking agent. Since excavation is not performed in the landfilling area, and since shallow groundwater or bedrock may prevail, the large quantities of soil required usually must be imported from off-site or hauled from other locations on-site.

Because filling proceeds above the ground surface, liners can be more readily installed at area fill operations than at trench operations. With or without liners, surface runoff of moisture from the sludge and contaminated rainwater should be expected in greater quantities at area fills.

There are three methods of area fill application (U.S. EPA, 1978). These are area fill mound, area fill layer and dike containment.

In area fill mound applications, it is recommended that the solids content of sludge received at the site be no more than 20%. Sludge is mixed with a soil bulking agent to produce a mixture that is more stable and has

greater bearing capacity. Appropriate bulking ratios may vary between 0.5 and 2 parts soil for each part of sludge. The exact ratio employed depends on the solids content of the sludge as received and the need for mound stability and bearing capacity.

The sludge/soil mixing process is usually performed at one location and the mixture hauled to the filling area. At the filling area, the sludge/soil mixture is stacked into mounds ~6 ft (1.8 m) high. Cover material is then applied atop these mounds in a minimum 3 ft (0.9 m) thick application. This cover thickness may be increased to 5 ft (1.5 m) if additional mounds are applied atop the first.

In area fill layer applications, sludge received at the site may be as low as 15% solids. Sludge is mixed with a soil bulking agent to produce a mixture that is more stable and has greater bearing capacity. Typical bulking ratios range from 0.25-1 part soil for each part sludge.

The mixing process may occur either at a separate sludge dumping and mixing area or in the filling area. After mixing the sludge with soil, the mixture is spread evenly in layers from 0.5-3 ft (0.15-0.9 m) thick. This layering usually continues for a number of applications. Interim cover between consecutive layers may be applied in 0.5-1 ft (0.15-0.3 m) thick applications. Final cover is from 2-4 ft (0.6-1.2 m) thick.

In diked containment applications, sludge is placed entirely above the original ground surface. Dikes are constructed on level ground around all four sides of a containment area. Alternatively, the containment area may be placed at the top of a hill so that the steep slope can be utilized for containment on one or two sides. Dikes would then be constructed around the remaining sides.

Access is provided to the top of the dikes so that hauling vehicles can dump sludge directly into the containment. Interim cover may be applied at

certain points during the filling, and final cover is applied when filling is discontinued. It is recommended that the solids content of the sludge be at least 20%.

Usually diked containment operations are conducted without the addition of soil bulking agents. Diked containments are relatively large with typical dimensions of 50-100 ft (15-30 m) wide, 100-200 ft (30-60 m) long and 10-30 ft (3-9 m) deep. In dike containment, the depth of the fill in conjunction with the weight of the sludge and cover fill results in much of the sludge moisture being squeezed into the surrounding dikes and into the floor of the containment. Thus, significant leachate emissions can be expected.

2.3. REVIEW OF SITE CONDITIONS AT OPERATED LANDFILLS

The previous discussions in this assessment were abstracted from the U.S. EPA Process Design Manual on Municipal Sludge Landfills (U.S. EPA, 1978). This document contains recommended guidelines for the operation of sludge landfills, though they are not always achieved at operating landfills. Also contained in the report are sludge and site descriptions of 22 operating or recently operated sludge landfills. Fifteen of these are sludge-only landfills. Tables 2-3 through 2-6 show the site and sludge characteristics of these sites. Depth to groundwater at these sites varies from 0-140 ft (42 m). The depth to groundwater at 12 sites appears to be within 20 ft (6 m). After trenching and/or filling, the depth to groundwater at 12 sites is <10 ft (3 m). Gravel is included in the description of soil type at four sites, and sand for seven; most of the sites are in areas of high rainfall [>20 in (51 cm)/year]; thus leachate generation can be expected. At some sites the aquifer beneath the landfill is used as a source of potable water. At most sites the sludge is exposed for <1 day, but at others it is as long as 60 days.

TABLE 2-3

Site Identification and Sludge Description*

Site Identification		Sludge Received							
				Quantity/Day			Quantity/Year		
		Treatment Process	Average Percent Solids	Cubic Meters	Wet Metric Tons	Dry Metric Tons	Cubic Meters	Wet Metric Tons	Dry Metric Tons
1.	Omaha, NE	CA, DE	28	181.2	195.0	50.8	51,763.4	55,792.8	14,515.2
2.	Fayetteville, AK	CA, DE	20	13.0	13.6	2.7	4557.0	4808.2	961.6
3.	Plattsburgh, NY	DE	12	123.1	127.0	16.3	44,805.6	46,267.2	5806.1
4.	Montgomery County, MD	CA, DE	22	344.1	362.9	78.9	91,140.3	96,163.2	20,865.6
5a.	Colorado Springs, CO TR(W)	CA, DE	22	46.6	49.9	10.9	16,821.2	18,144.0	3991.7
5b.	Colorado Springs, CO TR(N)	DI	3	415.2	426.4	12.7	107,808.6	110,678.4	3311.3
6.	Frederick, MD	DE	30	25.2	27.2	8.2	5856.8	6350.4	1905.1
7.	Fort Smith, AK	DI	30	22.9	24.5	7.3	4771.1	5171.0	1551.3
8.	Cleveland, OH	CA, DI, DE	26	383.8	408.2	89.8	139,921.8	148,780.8	32,659.2
9.	N. Kansas City, MO.	CA, DE	32	8.4	9.1	2.7	1299.8	1415.2	453.6
10.	Lewiston-Auburn, ME	CA, DE	21	49.7	51.7	10.9	13,074.7	13,608.0	2812.3
11.	Denver, CO	CA, DI, DE	15	636.9	604.2	90.7	191,150.0	199,584.0	30,300.4
12.	South Paris, ME	DE	14	63.5	58.1	8.2	19,803.1	18,144.0	2540.2
13.	Hartland, ME	CA, DE	19	50.5	52.6	10.0	13,151.1	13,698.7	2630.9
14.	Waterville, ME	CA, DE	20	37.5	39.0	8.2	13,762.8	14,333.8	2993.8
15.	Waukegan, IL	CA, DE	22	152.9	160.6	35.4	40,065.0	42,094.1	9253.4

*Source: U.S. EPA, 1978

1 cubic meter = 1.308 cubic yards
1 metric ton = 1.1023 short tons
CA = Chemical addition; DI = digestion; DE = dewatering; TR(N) = narrow trench; TR(W) = wide trench

TABLE 2-4

Site Design and Operation - Existing Conditions*

Site No.	Soil Type	Original Depth to Groundwater (meters)	Freezing (days/year)	Precipitation (cm/year)	Evaporation (cm/year)	Operational Method
1.	Sandy silt and clay	6.7	140	11	23.2	TR(W)
2.	Silty clay	3.0	90	20.5	23.2	TR(N)
3.	Sand, silt and clay	0.61-4.6	170	14.2	12.6	TR(W)
4.	Silty loam	1.8-11	90	15.7	18.5	TR(N)
5a.	Silt and clay	3.0	120	5.1	23.6	TR(W)
5b.	Silt and clay	3.0	120	5.1	23.6	TR(N)
6.	Clay	11.0	105	16.1	17.7	AF(L)
7.	Silty loam	2.4-3.0	75	17.3	25.2	TR(N)
8.	Sand, silt and clay	3.0-12.2	120	13.4	15.7	TR(W)
9.	Clay, silt and sand	3.7	105	13.8	23.2	TR(W)
10.	Sand, gravel and clay	0-0.30	160	17.3	12.2	AF(M)
11.	Clay, sand and gravel	3.0-42.7	240	5.5	23.6	AF(L)
12.	Sand	13.7-21.3	160	17.3	12.2	TR(N)
13.	Gravel and clay	6.1	160	17.3	12.2	TR(W)
14.	Gravel and sand	12.2	160	17.3	12.2	AF(M)
15.	Silty clay	9.44-12.2	140	12.6	15.4	TR(W)

*Source: U.S. EPA, 1978

1 meter = 3.281 feet
1 centimeter = 0.3937 inches
TR(N) = Narrow trench; TR(W) = wide trench; AF(L) = area fill layer; AF(M) = area fill mound

TABLE 2-5

Site Design and Operation - Site Preparation*

Site No.	Soil Type	Trench Width (meters)	Trench Depth (meters)	Trench Length (meters)	Trench Spacing (meters)	Sludge to Groundwater (meters)
1.	Sandy silt and clay	4.6	6.1	9.1	3.0	2.4
2.	Silty clay	0.6-0.9	3.0	variable	2.1-3.0	0.6
3.	Sand, silt and clay	106.7	6.1	335.3	15.2	0.6
4.	Silty loam	0.6	0.9	variable	0.6	0.9
5a.	Silt and clay	21.3	2.1	213.4	6.1-7.6	1.5
5b.	Silt and clay	0.6	3.0	45.7	4.6-6.1	0.6-0.9
6.	Clay	NA	NA	NA	NA	4.9
7.	Silty loam	0.76	2.4	variable	1.2-1.8	0.6
8.	Sand, silt and clay	12.2	1.5-1.8	137.2	1.5-3.0	1.2-10.4
9.	Clay, silt and sand	3.7	3.0	30.5	1.2-1.5	0.6
10.	Sand, gravel and clay	NA	NA	NA	NA	0-0.3
11.	Clay, sand and gravel	NA	NA	NA	NA	3.0-42.7
12.	Sand	1.8	1.8-2.4	12.2	1.2-3.0	11.3-18.9
13.	Gravel and clay	3.4-4.6	4.6	15.2	2.4	1.5
14.	Gravel and sand					12.2
15.	Silty clay	4.9-6.7	6.1	21.3	1.5-6.1	3.4-6.1

*Source: U.S. EPA, 1978

1 meter = 3.281 feet
NA = Not applicable

TABLE 2-6

Site Design and Operation - Site Operation*

Site No.	Soil Type	Sludge Application (cubic meters /hectare)	Fill Depth (meters)	Soil Used for Bulking	Soil Used for Cover	Cover Thickness (meters)	Sludge Exposure (days)
1.	Sandy silt and clay	27,395.2	6.1			0.6	30
2.	Silty clay	5479.0	2.9		X	0.6	3-4
3.	Sand, silt and clay	45,910.6	5.5				
4.	Silty loam	2408.9	0.6		X	0.9	1
5a.	Silt and clay	8879.8	1.2		X	1.2	1
5b.	Silt and clay	2172.7	2.1		X	1.5	4
6.	Clay	13,792.1	5.5	X	X	0.6-0.9	1
7.	Silty loam	4723.3	1.5		X	2.0	1
8.	Sand, silt and clay	7179.4	0.9		X	0.9-1.8	1
9.	Clay, silt and sand	20,026.9	4.0		X	2.1	21
10.	Sand, gravel and clay	26,450.5	4.6	X			4
11.	Clay, sand and gravel	17,003.9	5.3	X			1
12.	Sand	8502.0	1.5-2.1		X	0.9	1
13.	Gravel and clay	25,883.8	4.0		X	0.9	30
14.	Gravel and sand	16,248.2	3.7	X	X	0.3-0.6	60
15.	Silty clay	17,192.9	4.3		X	1.5	1

*Source: U.S. EPA, 1978

1 cubic meter = 1.308 cubic yards
1 hectare = 2.471 acres
1 meter = 3.281 feet

3. PATHOGENS

Raw sewage may contain a wide variety of pathogenic microorganisms. The pathogens include bacteria, viruses, protozoa, helminths and fungi, all of which can be expected to be present in raw, primary and secondary sludges. Pathogens currently of primary concern are listed in Tables 3-1 and 3-2. It should be recognized that the list of pathogens is not permanent, because as advances in analytical techniques and changes in society occur, new pathogens are recognized and the significance of well-known ones changes. Microorganisms are subject to mutation and evolution allowing for adaptation to changes in their environment. Thus, no risk assessment can be considered complete when dealing with microorganisms. As new organisms are discovered and a greater understanding of their ecology is developed, previous assumptions must be reevaluated.

3.1. BACTERIA

Members of the genus Salmonella are the most widely recognized enteric pathogens. Often associated with food and waterborne outbreaks of illness, they are responsible annually for 1-2 million incidents of disease in the U.S. population (Aserkoff et al., 1970). There are >1700 identified serotypes, many of which are able to infect both humans and animals. Salmonella species have been studied more than any other pathogenic bacteria found in sewage, and a good deal is known about their removal during sewage treatment and survival in the environment.

Shigella bacteria are responsible for ~3% of the reported diarrhea cases in the United States (APHA, 1975). The incidence of shigellosis in a community is clearly related to sanitation and water quality (Feachem et al., 1983). Four pathogenic species of Shigella are recognized, but little

TABLE 3-1

Bacteria and Parasites Pathogenic to Man That May Be Present in Sewage and Sludge*

Group	Pathogen	Disease/Symptom Caused
Bacteria	Salmonella (1700 types)	Typhoid, paratyphoid, salmonellosis
	Shigella (4 spp.)	Bacillary dysentery
	Enteropathogenic Escherichia coli	Gastroenteritis
	Yersinia enterocolitica	Gastroenteritis
	Campylobacter jejuni	Gastroenteritis
	Vibrio cholerae	Cholera
	Leptospira	Weil's disease
Protozoa	Entamoeba histolytica	Amoebic dysentery, liver abscess, colonic ulceration
	Giardia lamblia	Diarrhea, malabsorption
	Balantidium coli	Mild diarrhea, colonic ulceration
	Cryptosporidium	Diarrhea
Helminths	Ascaris lumbricoides (Roundworm)	Ascariasis
	Ancyclostoma duodenale (Hookworm)	Anemia
	Necator americanus (Hookworm)	Anemia
	Taenia saginata (Tapeworm)	Taeniasis
	Trichuris (Whipworm)	Abdominal pain, diarrhea
	Toxocara (Roundworm)	Fever, abdominal pain
	Strongyloides (Threadworm)	Abdominal pain, nausea, diarrhea
Fungi	Aspergillus fumigatus	Respiratory disease, otomycosis
	Candida albicans	Candidiasis
	Cryptococcus neopormans	Subacute chronic meningitis
	Epidermophyton spp. and Trichophyton spp.	Ringworm and athlete's foot
	Trichosporon spp.	Infection of hair follicles
	Phialophora spp.	Deep tissue infections

*Source: Gerba, 1983

TABLE 3-2

Enteric Viruses That May Be Present in Sewage and Sludge*

Viruses	Type	Disease/Symptom Caused
Enteroviruses:		
Poliovirus	3	Meningitis, paralysis, fever
Echovirus	31	Meningitis, diarrhea, rash, fever, respiratory disease
Coxsackievirus A	23	Meningitis, herpangina, fever, respiratory disease
Coxsackievirus B	6	Myocarditis, congenital heart anomalies, pleurodynia, respiratory disease, fever, rash, meningitis
New enteroviruses (Types 68-71)	4	Meningitis, encephalitis, acute hemorrhagic conjunctivitis, fever, respiratory disease
Hepatitis Type A (Enterovirus 72)	1	Infectious hepatitis
Norwalk virus	1	Diarrhea, vomiting, fever
Calicivirus	1	Gastroenteritis
Astrovirus	1	Gastroenteritis
Reovirus	3	Not clearly established
Rotavirus	2	Diarrhea, vomiting
Adenovirus	41	Respiratory disease, eye infections, gastroenteritis
Pararotavirus	unknown	Gastroenteritis
Snow Mountain Agent	unknown	Gastroenteritis
Epidemic non-A non-B hepatitis	unknown	Hepatitis

*Source: Gerba, 1983

data are available on their presence in domestic wastes and survival in the environment. There are no data available on Shigella destruction in most sludge treatment processes (Feachem et al., 1983). However, it is believed that Shigella destruction proceeds more rapidly than that of Salmonella or fecal indicator bacteria (Feachem et al., 1983).

Campylobacter bacteria are now recognized as a significant cause of enteric illness in animals and man. The species of most concern as an enteric pathogen in humans is Campylobacter jejuni. It is now thought to be at least as prevalent as Salmonella and Shigella pathogens and has been isolated from stools of 4-8% of patients with diarrhea (CDC, 1979). Outbreaks of enteric illness have been linked to fecally contaminated food and water. No information is currently available on the concentrations of this organism in sludge or its removal by sewage treatment processes.

Vibrio cholerae causes cholera, an acute enteritis characterized by sudden onset and rapid dehydration. The study of V. cholerae, atypical V. cholerae and non-01 V. cholerae has been attracting increasing attention in recent years because of several seafood-associated V. cholerae outbreaks along the Gulf Coast of the United States (Morris et al., 1981). Indeed, it has been suggested that it may be endemic in this region (Blake et al., 1980) and that the marine environment may be a natural reservoir for V. cholerae (Colwell, 1984). It appears that V. cholerae may survive for prolonged periods in wastewater, especially at low temperatures (Feachem et al., 1983). In a review of the literature, Feachem et al. (1983) were unable to find any reports on the occurrence of V. cholerae in sludge or during sludge treatment.

It is only in the last few years that Yersinia enterocolitica has been recognized as an etiological agent of acute enteritis. Yersiniosis occurs only sporadically in the United States and is transmitted from either infected animals or humans. Food and waterborne outbreaks have been documented (Feachem et al., 1983). This organism has been isolated from raw, digested and dewatered sludges (Metro, 1983).

Leptospira bacteria are excreted in the urine of domestic and wild animals. The bacteria enter municipal wastewater primarily from the urine of infected rats inhabiting sewers (Kowal, 1985). Leptospirosis, caused by Leptospira species, is uncommon in the United States (Kowal, 1985). The bacteria survive 2-4 days in the environment (Feachem et al., 1983). Leptospira organisms are rapidly destroyed during anaerobic sludge treatment and survival is probably <2 days (Feachem et al., 1983).

Although Escherichia coli is usually considered nonpathogenic, enterotoxigenic and enteropathogenic variants are responsible for numerous outbreaks of enteritis. Several studies in different parts of the world have indicated that E. coli is a significant cause of bacterial diarrhea, and food and waterborne outbreaks of E. coli-caused illness have been documented (Feachem et al., 1983).

Because indicator bacteria are easy to study and occur abundantly in sewage and sludge, a great deal is known about the removal of coliform, fecal coliform and fecal streptococci by sewage treatment processes and their survival in the environment. While not normally considered "frank" pathogens, they may cause disease especially in compromised or immunosuppressed individuals. In many, if not most situations, it is believed that members of these indicator groups are as resistant to treatment removal and survive in the environment similar to that of the

enteric bacterial pathogens, although it is clearly not the situation in all cases (Feachem et al., 1983).

3.2. VIRUSES

More than 100 different virus types may be present in raw sewage (see Table 3-2). The list of pathogenic human enteric viruses, which could be present in sewage, increased at the rate of 1.3/year from 1972-1983. There are obviously many more viruses yet unrecognized that could be present in domestic wastes. Most of the knowledge on viruses in sewage is limited to those associated with gastroenteritis. Exceptions are certain enteroviruses, which are associated with a wide variety of diseases, and adenoviruses, which may cause eye infections. Enteroviruses are often associated with more serious illnesses such as hepatitis, meningitis, myocarditis and paralysis (see Table 3-2).

The most commonly studied enteric viruses in sewage and sludge are the enteroviruses, which include polioviruses, coxsackie A and B viruses, echoviruses, hepatitis A virus and other recently classified enterovirus types. Several new, presently unclassified enteroviruses, which have been responsible for foodborne outbreaks of illness in Australia, have been recently isolated in cell culture (Grohmann, 1985). While many of the enterovirus infections such as those caused by poliovirus may be asymptomatic, symptomatic infections may be as high as 95% during outbreaks of hepatitis (Lednar et al., 1985). A great deal of information is available on the removal of enteroviruses by sewage treatment, and many studies have been conducted on their occurrence in sludges (Leong, 1983).

Rotaviruses are now recognized as a major cause of childhood gastroenteritis, sometimes resulting in dehydration and death in infants and adults (Gerba et al., 1985). Several waterborne outbreaks have been docu-

mented (Gerba et al., 1985; Williams and Akin, 1986) and the virus has been isolated from sewage sludges (Gerba, 1986a).

The Norwalk virus has been demonstrated to be the cause of numerous waterborne outbreaks of epidemic gastroenteritis (Gerba et al., 1985). Since methods have not been developed for its isolation in cell culture, its occurrence and concentration in sewage and sludges is unknown. Astroviruses, caliciviruses, coronaviruses, pararotaviruses and several other Norwalk-like agents have been associated with human gastroenteritis, but little is known about them. Laboratory methods are currently not available to study most of these agents and they await further characterization.

Adenoviruses primarily cause respiratory infections and eye infections, although several new types have been found associated with gastroenteritis (Gary et al., 1979).

An epidemic non-A non-B hepatitis virus, which has caused large-scale outbreaks of waterborne disease in Asia and Africa, has recently been recognized (Gerba et al., 1985).

3.3. PROTOZOA

In the past, little attention has been given to the presence of parasites in sewage because of the popular impression that the prevalence of parasite infection in the United States is low (Larkin et al., 1976). However, the continuing occurrence of waterborne outbreaks of giardiasis and the resistance of cysts to disinfection indicate that they deserve serious consideration (Erlandsen and Meyer, 1984).

Of the common protozoa that may be found in sewage, only four species are believed to be of major significance for transmission of disease to humans (see Table 3-1). All four, Entamoeba histolytica, Giardia lamblia, Balantidium coli and Cryptosporidium sp. cause mild to severe diarrhea.

Waterborne outbreaks of disease for all of these agents are known to have occurred. G. lamblia is now the agent most commonly associated with waterborne disease outbreaks in which an agent can be identified (Craun, 1984). Cryptosporidium sp. has only recently been recognized as a pathogen in man. It infects both animals and man and is apparently a cause of traveler's diarrhea (Sterling, 1986) and gastroenteritis worldwide. A waterborne outbreak of cryptosporidiasis was recently reported in Texas (D'Antonio et al., 1985), and the pathogen was also recently isolated from domestic sewage effluents (Musial, 1985) and sludge (Gerba, 1986a).

Limited information is available on the occurrence of protozoa in sewage, but even less is known about their occurrence in sludges.

3.4. HELMINTHS

A wide variety of helminths and their eggs may occur in domestic sludges. Helminths are worms that include nematodes (roundworms) and cestodes (tapeworms). Those of primary concern are listed in Table 3-1. Many common helminths are pathogenic to domestic animals, such as cats and dogs. Helminths have been identified in domestic wastewater and sludge, and Reimers et al. (1981) have found Ascaris, Trichuris and Toxocara helminth eggs in municipal wastewater sludge in both the southeastern and northern United States.

Ascariasis is a helminthic infection of the small intestine in humans by the roundworm, Ascaris lumbricoides. About 85% of the infections are asymptomatic, although the presence of even a few worms is potentially dangerous (Feachem et al., 1983). Large numbers of worms may cause digestive and nutritional disturbances, abdominal pain and damage to internal organs. The prevalence of ascariasis in the United States was estimated to be ~4 million in 1972 (Warren, 1974).

Ascaris eggs tend to become concentrated in the sludge during sewage treatment and their removal by sludge treatment has been studied (Feachem et al., 1983).

Trichuriasis is an infection in humans caused by the whipworm, Trichuris trichiura. Trichuriasis is a helminthic infection of the large intestine and cecum. Most infections in adults are asymptomatic, but there may be slight abdominal pain and diarrhea. Trichuris eggs, like Ascaris eggs, tend to settle in primary and secondary sedimentation tanks and are, therefore, concentrated in the sludge from sewage treatment plants. The fate of Trichuris eggs during sludge storage, digestion or composting is believed to be similar to that for Ascaris eggs, but Trichuris eggs are probably eliminated somewhat earlier during these processes (Feachem et al., 1983).

Ancyclostomiasis is an infection of the small intestine with one of the two species of hookworms, Necator americanus or Ancyclostoma duodenale. Ancyclostomiasis is frequently symptomless. When it does produce illness and constitutes a public health problem, the most important features are anemia and debility. Because of the low incidence of hookworm in the United States, only low numbers have been found in sludge. Hookworm eggs and larvae are less resistant to sludge treatment processes than Ascaris eggs (Feachem et al., 1983). Problems could arise if raw or inadequately treated sludges are applied to pastureland, since once in the soil the eggs will hatch and produce infective larvae.

Taenia saginata and T. solium, the beef and pork tapeworms, live in the intestinal tract where they may cause abdominal pain, weight loss and digestive disturbances. The infection arises from eating incompletely cooked meat containing the larval stage of the tapeworm, rather than from a wastewater-contaminated material. Humans serve as the definitive host, harboring

the adult. The eggs are passed in the feces, ingested by cattle and pigs (intermediate hosts), hatch, and the larvae migrate into the tissues, where they develop to the cysticercus stage. The hazard is then principally to livestock grazing on sludge application sites. Taenia eggs are concentrated in sewage sludge and may survive for prolonged periods after land disposal (Feachem et al., 1983). Taenia eggs may not be completely destroyed by all sludge treatment processes (Feachem et al., 1983). An investigation of an outbreak of T. saginata near Tucson, AZ, revealed that cattle became infected while grazing on a farm pasture irrigated with primary sewage effluent (Slonka et al., 1975). Pastureland fertilized with municipal sludge was implicated in a T. saginata outbreak in Virginia (Hammerberg et al., 1978).

3.5. FUNGI

Fungi are usually considered to be of minimal health risk in the application of municipal sludge. The pathogenic fungi listed in Table 3-1 can all be recovered from municipal sludge (WHO, 1981).

These fungi can be divided into two groups, the yeasts and the filamentous molds. The yeasts include Candida albicans and other Candida spp., Cryptococcus neoformans and Trichosporon spp., whereas the filamentous mold fungi include the various species of Aspergillus, especially A. fumigatus, Epidermophyton spp., Phialophora spp. and Trichophyton spp. These fungi have been reported to be present in sewage and in all stages of sludge treatment (WHO, 1981); because of their environmentally ubiquitous existence, it is difficult to evaluate their significance to public health. The World Health Organization's Working Group on Sewage Sludge Applied to Land: Health Implications of the Microbial Content (WHO, 1981) emphasized that because of their prevalence in nature, even if the sludge were treated by pasteurization, fungi will recontaminate the sludge.

Aspergillus fumigatus is one of the most prevalent fungi in municipal compost. This opportunistic pathogen may cause upper respiratory tract infections in man (WHO, 1981). Since composted sludge is not buried in landfills, this fungus is not considered in this document.

4. EXPOSURE PATHWAYS

The possible exposure pathways by which infectious microorganisms may reach humans from the operation of sludge landfills are shown in Figure 4-1. Exposure to one or more routes of transmission is dependent on significant numbers of microorganisms being present to result in infection. It is not inconceivable that some microorganisms during sludge disposal follow all of the routes illustrated in Figure 4-1. However, it is unlikely that significant numbers are transmitted by all of the pathways.

Personnel may be exposed through direct contact with the sludge or through exposure to aerosols generated during burial. Aerosols could be transported downwind to areas distant from the disposal site. Aerosols containing viable microorganisms may also contaminate clothing and equipment directly. Microorganisms may leach from the buried sludge with infiltrating water to contaminate the groundwater. Exposure of the sludge to the surface would result in the generation of runoff, which would transport sludge particles to nearby surface waters. It is also possible that if the site becomes saturated with water, surface leachate contamination will occur. Burrowing animals could come into contact with the buried sludge and birds would be exposed to the sludge before burial. These animals could serve to transport sludge material off-site or expose it to the surface. Translocation of viruses from the subsurface plant roots to the aerial parts of the plant is another potential pathway.

4.1. AEROSOLS AND DIRECT CONTACT

Many enteric microorganisms can effectively be transmitted by aerosols. In fact, the infectious dose by the aerosol route may be less than by the

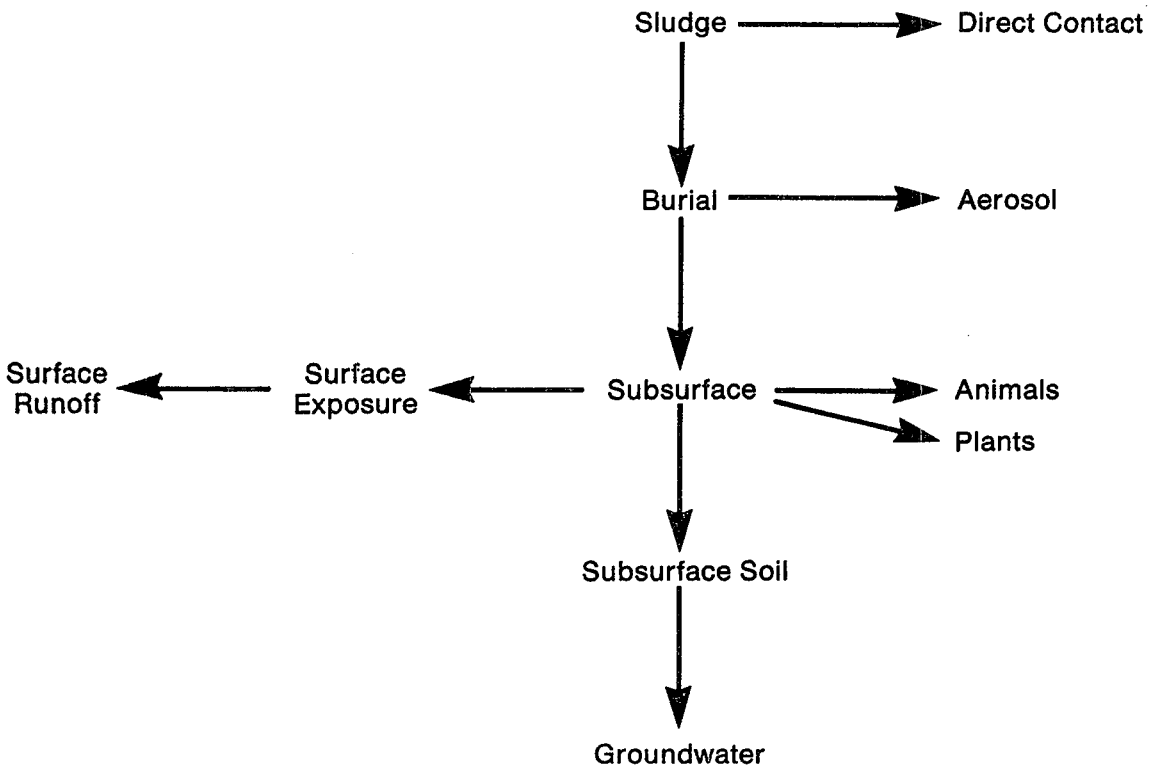


FIGURE 4-1

Pathways of Microbial Transport from Sludge Landfills

ingested route for some organisms. Aerosols of enteric organisms are generated during sewage treatment and during the spraying of sewage effluents and sludges onto land (Pahren and Jakubowski, 1980). The organisms in such aerosols can be transmitted by inhalation or the settling of the organisms onto surfaces with which humans come into contact.

The number of microorganisms generating aerosols depends on the type of sludge disposed, method of application and burial, and number of microorganisms present in the sludge. The greatest amount of aerosol generation occurs during application of sludges with a low solids content applied as slurries. Dumping of sludges from trucks into trenches and area fills also generates aerosols as the sludge impacts the ground. Aerosols may be more contained during disposal into trenches because of protection from wind scour. The greatest chance for transport of aerosols off-site occurs with area fill operation. Some aerosol generation occurs during burial of the sludge. Greater numbers of pathogenic organisms form aerosols during disposal of primary sludges than treated sludges.

If wind velocities at a site are great enough, suspension of the sludge particles could occur. Most sludges would not be easily resuspended because of their moisture content and tendency to mat as they dry. Dried sludges, however, may be very light and fine in texture and, therefore, easily resuspended. Wind speeds great enough to resuspend sludge would be unlikely most of the time even at sites with a wind-power potential (U.S. EPA, 1986). Thus, wind data coupled with the operating times of $\leq$50% without cover suggest that for windy sites, the winds will attain speeds capable of suspending the sludge from the working face for brief periods of time. Such resuspensions could be controlled by requiring placement of daily soil cover over landfilled sludges (U.S. EPA, 1986).

The possibility of health risks from public and occupational exposures to these aerosols has been discussed extensively (Pahren and Jakubowski, 1980). Several studies have dealt with the measurement of aerosols from activated sludge treatment plants and spray application of wastewater to land. These studies included aerosol monitoring and attempted to examine health effects in populations either working at the site or living nearby. However, epidemiological studies have not produced conclusive results as to the impact of such aerosols on human health (Pahren and Jakubowski, 1980).

The use of tank trucks and high-volume spray guns for application of liquid sludges is much more likely to generate significant microbial aerosols than landfilling. In a study of aerosols generated during the land application of liquid sludge, Sorber et al. (1984) reported that microbial aerosol concentrations are less than those at wastewater spray application sites, and no significant health effects should occur in individuals located >100 m downwind of the sludge application site.

In summary, some aerosolization of pathogenic organisms will occur during landfilling of sludges. Occupational exposure to workers will occur with a possible risk of disease transmission. Through proper management and the use of a buffer zone, significant microbial aerosols should not occur off-site.

4.2. SURFACE WATER AND RUNOFF

During sludge landfill operations, it is normal practice to bury the sludge under several feet of earth at the end of each day. Even in those operations where the sludge may be exposed for several days, the sludge is contained in trenches or pits that limit exposure of the sludge to surface runoff. Thus, exposure by this route is insignificant unless the sludge becomes exposed by removal of the soil covering. However, if the

site becomes saturated with water through rising water tables or flooding, leachate from the buried sludge could reach the surface and be transported from the site with surface runoff.

If suspended sludge particles or leachate leave the site, the material could contaminate surface recreational areas, irrigated food crops and drinking waters and pose a threat to human health.

Operating procedures at sludge landfills require control of runon and runoff from the working face with drainage ditches. In addition, since the working face is below grade for the surrounding areas, all trench or pit fills contain runoff by design. However, this is not the case for area or canyon fills; provisions must be made to contain drainage in the down-gradient direction in these fills. Based on the assumption of good operating practices, runoff becomes a part of the groundwater pathway or is eliminated. The precipitation that runs off of the working face will collect at the foot or in a drainage control ditch where it will either percolate into the soil, be used for dust control or be routed to treatment. Therefore, the methodology in this assessment does not include an independent surface runoff pathway.

4.3. PLANT AND ANIMAL

Potentially, plant roots and burrowing animals could come into contact with the buried sludge. In addition, birds (seagulls) could become exposed to the sludge before burial. Translocation of viruses from the subsurface plant roots to the aerial parts of the plants has been observed (Murphy and Syverton, 1958; Ward and Mahler, 1982), but only when grown in hydroponic culture or when the roots were cut. Ward and Mahler (1982) concluded that it was unlikely that viruses penetrate the intact surfaces of roots. Birds whose feet or other body parts become contaminated could then carry the

contamination to drinking water reservoirs. However, transport of significant amounts of pathogenic microorganisms by this route appears unlikely.

4.4. GROUNDWATER

Contamination of groundwater and use of that groundwater for domestic purposes appears the most likely route of significant human exposure from sludge burial. Many of the sludge landfills in the United States are operated over aquifers that are used as potable sources. A review of operating landfill sites (see Section 2.3.) indicates that many are constructed within a few meters of the groundwater table. Therefore, this pathway is considered in the greatest detail in this assessment.

5. EXPECTED CONCENTRATIONS OF PATHOGENS IN SLUDGE

The concentrations and types of pathogens in sludges depend on two principal factors: the incidence of infection within a community and the type of treatment the sludge receives. Season, climate and sanitation are major factors that will determine the pathogenic load a wastewater treatment plant receives. Various sludge treatment processes, such as anaerobic digestion and dewatering, act to reduce the numbers of some of the pathogens initially present.

There are two main types of wastewater sludge: primary and secondary. Primary sludge is obtained after gravity sedimentation of solids in raw wastewater. This process removes ~60% of the total suspended solids from sewage and results in a semisolid product that typically contains ~5% solids by weight and has a pH of ~6. Both the percentage of solids and pH are dependent on the characteristics of the specific treatment plant and the source of sewage.

Secondary sludges are obtained from wastewater treated by any of a number of secondary processes. Most treatments are biological, such as the activated-sludge process, trickling filters and rotating biological contactors. Secondary sludges obtained following biological treatment of wastewater typically have low percentages of solids and may be thickened by flotation, centrifugation or other means. Secondary sludges are often combined with primary sludges for further treatment, but they may also be processed separately.

Prior to disposal, both primary and secondary sludges must be treated to reduce volatile solids (stabilization) and dewatered.

Sludges are stabilized to eliminate odor problems, reduce pathogen numbers and prevent decomposition under unwanted conditions. Sludges may also receive further treatment to reduce pathogens by methods such as composting, heat drying, heat treatment (pasteurization) and γ-irradiation.

5.1 PATHOGEN CONCENTRATIONS IN RAW SLUDGES

Most microbial species contained in raw sewage are concentrated in sludge during primary sedimentation. Enteric viruses have too little mass to settle alone but, because of their strong binding affinity to particulates, they are also concentrated in sludge.

The numbers shown in Table 5-1 represent typical, average values that have been detected by a number of investigators. Different sludges may contain significantly more or less of any organism as determined primarily by the sewage from which the sludge was derived. The quantities of pathogenic species will be especially variable, depending on which are circulating in a community at any particular time. Indicator organisms are normally present in fairly constant amounts. Because the concentrations determined in any study are dependent on particular assays to detect each microbial species, these concentrations are only as accurate as the assays themselves. This point is especially relevant in regard to viruses for which only a small percentage is normally detected by even the best procedures.

5.2. PATHOGEN CONCENTRATIONS IN SECONDARY SLUDGES

The secondary sludges of concern in this report are produced following biological treatments of wastewater. Microbial populations in sludges following these treatments will depend on the initial concentrations in the wastewater, the die-off or growth during treatments and the association of these organisms with sludge (Ward et al., 1984). Some treatments, such as the activated-sludge process, have a deleterious effect on enteric microbial

TABLE 5-1

Densities of Microbial Pathogens and Indicators in Primary Sludges*

Type	Organism	Density (number/g dry weight)
Virus	Various enteric viruses	10^2-10^4
	Bacteriophages	10^5
Bacteria	Total coliforms	10^8-10^9
	Fecal coliforms	10^7-10^8
	Fecal streptococci	10^6-10^7
	Salmonella sp.	10^2-10^3
	Clostridium sp.	10^6
	Mycobacterium tuberculosis	10^6
Parasite	*Ascaris* sp.	10^2-10^3
	Trichuris vulpis	10^2
	Toxocara sp.	10^1-10^2

*Source: Ward et al., 1984

species. Concentrations of viral and bacterial pathogens have been shown to be reduced by activated-sludge treatment. Even so, the ranges of concentrations in secondary sludges obtained following this and most other secondary treatments are usually not significantly different from those of primary sludges. Examples are shown in Table 5-2.

5.3. PATHOGEN CONCENTRATIONS AFTER STABILIZATION, DEWATERING AND DISINFECTION

Anaerobic digestion is probably the most common method of sludge stabilization practiced in the United States (Ward et al., 1984). It consists of the degradation of complex organic substances by microorganisms in an environment devoid of free oxygen. Primary and secondary sludges are fed either continuously or on an intermittent basis into an airtight container. Heat is normally supplied from an exogenous source to energize indigenous anaerobic microorganisms. Retention times are variable. The process is usually conducted in the absence of air at residence times ranging from 30-60 days at 20°C to 15 days at 35-55°C, with a volatile solids reduction of at least 38% (U.S. EPA, 1974).

Certain conditions within an operating digester are normally quite constant. The pH is between 6.5 and 7.5 and the water content is generally ~95%. However, the temperatures at which the digesters are maintained can be quite variable. This is probably the most important operational parameter affecting pathogen survival (Ward et al., 1984).

In contrast to other pathogens indigenous to sludge, bacteria are free living and can multiply outside of their hosts. Thus, it is possible for bacterial numbers to increase. However, the environment does not appear to be conducive to the growth of enteric bacterial pathogens (Ward et al., 1984).

TABLE 5-2

Densities of Pathogenic and Indicator Microbial Species in Secondary Sludges*

Type	Organism	Density (number/g dry weight)
Virus	Various enteric viruses	3×10^2
Bacteria	Total coliforms	7×10^8
	Fecal coliforms	8×10^6
	Fecal streptococci	2×10^2
	Salmonella sp.	9×10^2
Parasite	*Ascaris* sp.	1×10^3
	Trichuris vulpis	$<10^1$
	Toxocara sp.	3×10^2

*Source: Ward et al., 1984

Mesophilic digestion of sludge can result in 1 to 2 orders magnitude of loss of pathogenic bacteria and viruses (Ward et al., 1984) (Table 5-3). Little inactivation of parasites such as Ascaris lumbricoides can be expected under these conditions. However, temperature is a major factor in the survival of these organisms during this process. Thus, sludge undergoing thermophilic digestion may largely be free of pathogens.

Aerobic digestion is conducted by agitating sludge with air or oxygen to maintain aerobic conditions at residence times ranging from 10-60 days (U.S. EPA, 1974).

Aerobic digesters are fed with the same types of sludges as anaerobic digesters. Therefore, they receive the same pathogen load. They are normally operated at ambient temperatures. However, the possibility of pathogen die-off from high temperatures is reduced relative to anaerobic digestion where temperatures are normally held at ~35°C.

There is a paucity of data regarding pathogen inactivation during aerobic digestion of sludge. Ward et al. (1984) estimated that reductions in pathogens would be in the same order of magnitude observed for anaerobic digesters (see Table 5-3).

Another type of process used for stabilization of sludges disposed of in landfills is chemical treatment with lime. This process differs from biological forms of stabilization in that it does not affect the availability of food for microbial growth. Lime treatment is also used to aid in the dewatering of sludge. The high pH during this process affects microbial survival. The pH must be sufficiently high for a long enough period if the treatment is to be effective for inactivating viral and bacterial pathogens. It is well established that the initial pH of sludge is dependent on the amount of lime added but that the pH decreases significantly over a period of hours.

TABLE 5-3

Summary of Microbial Reduction During Sludge Treatment[a]

Treatment	Reduction[b]		
	Bacteria	Viruses	Parasites
Anaerobic digestion[c]	1-2	1	0
Aerobic digestion	2-2	1	0
Composting	2-3	2-3	2-3
Air drying[d]	2-3	1-3	1-3
Lime stabilization	2-3	3	0

[a]Source: Ward et al., 1984

[b]Scale: 0 -- <0.5 orders of magnitude
1 -- 0.5-2 orders of magnitude
2 -- 2-4 orders of magnitude
3 -- >4 orders of magnitude

[c]Mesophilic temperatures are assumed.

[d]Effects depend on moisture levels attained.

Substantial inactivation of viruses may occur during lime treatment. Parasite ova are resistant to high pH, and most probably will survive lime treatment. Bacteria are rapidly inactivated at pH 12 but, because the pH decreases to levels suitable for bacterial growth, their numbers increase with time. Bacterial pathogens have not been found in sufficiently limed sludge, but their regrowth can be anticipated under proper conditions (Ward et al., 1984).

Air drying and dewatering may also result in pathogen reduction in sludges (see Table 5-3). The concentration of pathogens observed in stabilized sludges is shown in Table 5-4.

Other nonconventional treatment or disinfection processes such as heat drying, pasteurization, heat treatment and γ-irradiation also act to reduce the numbers of pathogens present in sludge before disposal. Their effectiveness on pathogen removal is discussed by Ward et al. (1984).

TABLE 5-4

Concentrations of Pathogens and Indicators in Digested Sludges*

Organism	Type of Stabilization		Average of All Types of Digested-Dewatered Sludge (90 samples positive)	Percent Viable
	Anaerobic (g dry weight)	Aerobic		
Enteroviruses	0.2-210	0-260		
Rotaviruses	14-485	ND		
Salmonella spp.	3-10^3	3		
Total coliforms	10^2-10^6	10^5-10^6		
Fecal coliforms	10^2-10^6	10^5-10^6		
Shigella sp.	20	ND		
Yersinia enterocolitica	10^5	ND		
Ascaris sp.	-	-	9.6	45
Trichuris sp.	-	-	2.6	48
Toxocara sp.	-	-	0.7	52

*Source: Data compiled from Sagik et al., 1980; Reimers et al., 1981; Metro, 1983; Kowal, 1985; Badawy, 1985; Gerba, 1986a

ND = No data

6. SURVIVAL CHARACTERISTICS OF PATHOGENS

Most pathogenic microorganisms have a finite lifetime in the environment once they have left the host organism. However, under the proper conditions they may actually grow and increase in numbers. For a risk assessment of pathogens in landfills it is necessary to predict the persistence of these organisms in the sludge-soil and groundwater environments.

Information on pathogen survival in sludge landfills and leachate is essentially nonexistent. However, studies with a laboratory lysimeter suggest that total coliforms may persist for at least 100 weeks in buried sewage sludge (Donnelly and Scarpino, 1984). The literature on pathogen survival in water, sludge and soil was reviewed to determine significant factors controlling survival and the use of models for predicting pathogen die-off or inactivation.

6.1. SLUDGE AND SOIL

6.1.1. Viruses. Solids-associated viruses in sludge have been shown to be infectious, well protected and able to survive longer in the environment than free-living viruses in water and wastewater (Gerba, 1984a). Most available information describes viral survival in soil alone, rather than survival in a sludge-soil matrix. Only a few studies have been conducted investigating virus survival in digested sludge applied to soil. These studies generally indicate that viral inactivation in sludge can be a slow process. In Boulder, CO, enteroviruses were isolated from sludge-soil samples taken 30 days after subsurface sludge injection (Metro, 1983). In sludge-soil samples from a subsurface injection site in Butte, MT, viruses were recovered 6 months after application (Moore et al., 1977). At a site where sludge was applied to a forest plantation, enteroviruses were detected

in the soil for as long as 21 weeks after application (Jorgensen and Lund, 1985). In soil flooded with inoculated sewage sludge, poliovirus 1 was found to survive for at least 96 days during the winter and 36 days during the summer (Larkin et al., 1976). Unfortunately, none of these studies provide quantitative data, which could be used to aid in predicting rates of virus inactivation.

Soil moisture and temperature are the main factors determining virus survival in soils, although the nature of the soil may also play a role (Hurst et al., 1980a). In a study using seeded effluent applied to soil columns, a 99% die-off of poliovirus 1 occurred in clay soil after 10 days at 30°C. At 4°C, a comparable die-off did not occur even after 134 days. At 15% moisture, poliovirus survival seemed to be optimal (Duboise et al., 1979). A similarly designed study demonstrated prolonged survival of poliovirus 1 at low temperature and soil moisture of 15-25% (Moore et al., 1977). Only a 90% decrease in virus was seen in 3 months at 4°C, and in 1 month at 20°C. Under warm (30°C) dry conditions, inactivation occurred in 1 week. Naturally occurring enteroviruses have been isolated from soils beneath a sludge disposal site in Denmark (Jorgensen and Lund, 1985) and at several sites where land application of domestic sewage was practiced (Hurst et al., 1980b; Goyal et al., 1984). There are several recent extensive review articles concerning virus survival in soil and groundwater systems (Sobsey, 1983; Vaughn and Landry, 1983; Gerba and Bitton, 1984).

Of all the enteric viruses, hepatitis A virus (HAV) (enterovirus type 72) may be the most resistant to thermal inactivation and inactivation in general in the environment (Peterson et al., 1978; Siegel, 1982). HAV has recently been shown to survive for prolonged periods of time in sewage effluents and soils (Hazard and Sobsey, 1985; Sobsey, 1985). The decay

rates observed by Sobsey (1985) for HAV, poliovirus and echovirus in water and soil are shown in Table 6-1. These preliminary results suggest that HAV may be substantially more resistant to inactivation in soil and water than are the other enteroviruses.

Hurst et al. (1980b) determined the inactivation rates of poliovirus 1 and echovirus 1 buried in basins of sandy soil flooded with sewage to be 0.04 and 0.15 $\log_{10}$ day^{-1} when the basins were flooded (that is, saturated flow). When the basins were dry, decay rates ranged from 0.11-0.52 $\log_{10}$ day^{-1}. Since decay rates are dependent on soil moisture loss, virus inactivation could be expected to be greater in unsaturated conditions where soil moisture loss is significant. A number of other investigators have studied the decay rates of viruses in soils, but since the information was derived from soils subject to drying, its usefulness in predicting virus decay in the deep subsurface is subject to question (Reddy et al., 1981).

Viruses adsorbed to soil and/or sludge can be expected to survive longer than when freely suspended in the groundwater. Apparently, adsorbed viruses are protected against inactivation (Liew and Gerba, 1980; Gerba, 1984b). Information on virus survival in groundwater has only become available in the last few years. Enteroviruses have been isolated from groundwater at numerous sites where land application of wastewater is practiced (Keswick and Gerba, 1980). Stramer (1984) observed that poliovirus 1 survived in groundwater over 100 days after leaving a septic tank.

6.1.2. Bacteria. Bacterial die-off is influenced by many of the same factors as virus inactivation with the addition of the availability of nutrients playing a role. Temperature, pH, moisture and nutrient supply have the greatest impact on enteric bacterial survival (Gerba et al., 1975).

TABLE 6-1

Comparative Die-Off of Enterovirus in Water and Soil[a]

Type of Water or Soil	Decay Rates (k day^{-1})					
	Hepatitis A		Poliovirus 1		Echovirus 1	
	5°C	25°C	5°C	25°C	5°C	25°C
Groundwater	0.006	0.21	ND	ND	ND	ND
Primary sewage effluent	0[b]	0.024	0.0089	0.143	0.012	0.214
Groundwater in Kaolinite	0[c]	0.0045	ND	ND	ND	ND
Groundwater in Corolla	0.0045	0.036	ND	ND	ND	ND
Groundwater in FM	0.009	0.143	ND	ND	ND	ND
Primary sewage in Corolla	0.0012	0.0178	0.006	0.089	0.006	0.071
Primary sewage in FM	0[b]	0.02	0.006	0.066	0.006	0.071

[a]Source: Sobsey, 1985

[b]No measurable inactivation after 84 days

[c]No measurable inactivation after 56 days

ND = No data

FM = Loamy soil

Corolla = Coarse sand

Antagonism by competing microflora may play a significant role, but it is difficult to quantify. Like most enteric microorganisms, lower temperatures increase survival time in soil (Crane and Moore, 1984), although freezing and thawing conditions are detrimental (Kibbey et al., 1978). Extremes in pH are also detrimental to bacterial survival (Kibbey et al., 1978; Hudson and Fennel, 1980). Generally, a near neutral pH environment favors extended bacterial survival (McFeters and Stuart, 1972). Beard (1940) found that Salmonella typhosa survived best between pH 6.5-8 in soils.

Moisture effects in soil systems are of major importance in bacterial decline. Kibbey et al. (1978) found that bacterial survival rates for Streptococcus faecalis and Salmonella typhimurium increased with increasing moisture content of the soil at several different temperatures. When sludges are buried, soil moisture loss is probably minimized (Crane and Moore, 1984). Bacterial survival apparently would be greatest under saturated conditions (Boyd et al., 1969; Kibbey et al., 1978). Of all these factors, temperature is the most easily quantified. A review of the literature by Reddy et al. (1981) indicates that die-off rate approximately doubles with each 10°C rise in temperature between 5 and 30°C. Die-off rate coefficients could be adjusted for temperature by using the following equation:

$$kT_2 = kT_1 \cdot F_T$$

where

$F_T = \theta^{T_2-T_1}$
kT_2 = die-off rate adjusted for temperature T_2
kT_1 = die-off rate measured at temperature T_1
θ = temperature correction coefficient
T = temperature (°C).

In the studies reviewed by Reddy et al. (1981) the temperature correction coefficient (θ) ranged from 1.02-1.17, with an average value of 1.07 ± 0.05 (Table 6-2).

The nutrient supply and organic matter contained in the soil and percolating water also affect the rate of bacterial die-off. A major reason for enteric bacterial die-off outside of the host intestinal tract is probably the inability of these organisms to lower their metabolic requirements to a situation of lower nutrient availability (Klein and Casida, 1967). Mallman and Litsky (1951) felt that organic content present in sludge enhanced bacterial survival. The survival of fecal coliforms is greatly extended in organic soils over that observed in mineral soils (Tate, 1978), and regrowth of S. typhimurium and E. coli has been observed in buried feces (Temple et al., 1980).

Of all the pathogenic bacteria, Salmonella survival has been studied most extensively (Feachem et al., 1983). Salmonella bacteria can survive in animal slurries, sludges and soils for periods of many months when conditions are ideal (that is, high moisture and low temperatures). Salmonellae in sludge applied to arid land in summer persisted for 6-7 weeks (Watson, 1980). Hess and Breer (1975) reported that salmonellae on grass treated with sludge could survive for $\leq$16 months in the climate of Switzerland, but most reported times are shorter than this. Salmonella organisms can multiply vigorously in sterilized sludge or slurry, but under natural conditions growth is limited or strongly inhibited by the activity of other microflora (Findlay, 1973).

Although the shigellae are among the most important pathogenic enteric bacteria, their presence and persistence in the environment have been studied far less than is the case for E. coli and the Salmonella. In clean waters, survival times are typically <14 days at warm temperatures (>20°C),

TABLE 6-2

Temperature Correction Coefficients for the Survival of Pathogens and Indicator Organisms in Soil and Water Systems*

Type of Organism	Temperature Range (°C)	Temperature Correction Coefficient (θ)
Fecal coliforms	15-21	1.08
Escherichia coli	5-10	1.09
	10-15	1.17
	15-20	1.15
	20-25	1.07
Aerobacter aerogenes	10-20	1.02
Salmonella typhimurium	10-20	1.05
Enterobacter aerogenes	10-20	1.03
Streptococcus faecalis	4-10	1.02
	10-35	1.03
	25-37	1.06
Average	4-37	1.07±0.05

*Source: Reddy et al., 1981

whereas the bacteria may survive for a few weeks below 10°C (Feachem et al., 1983). Interestingly, McFeters et al. (1974) found that shigellae died more slowly in wellwater at 9-12°C than the fecal bacterial indicators, salmonellae or Vibrio cholerae. No studies could be located on the survival of Shigella organisms in soils or sludge. A review of the literature on Shigella survival in the environment by Feachem et al. (1983) suggests that at temperatures >30°C, Shigella survival is less than Salmonella.

V. cholerae appears capable of surviving for 4-10 days in soils moistened with sewage at 20-28°C (Gerichter et al., 1975). Data are not available on the survival of V. cholerae in sewage sludges. Although the traditional view has been that V. cholerae does not survive for prolonged periods in the environment, more recent studies have suggested that prolonged survival and regrowth are possible under certain conditions (Feachem et al., 1983). Based on a review of the literature, Feachem et al. (1983) calculated t_{90} values in hours for V. cholerae in various types of waters (Table 6-3). This review suggests that V. cholerae exhibits longer survival in wellwater and seawater than in fresh surface waters and sewage. In general, though, it would appear that above 30°C V. cholerae survival would be less than that of Salmonella.

Little is known about the occurrence and survival of Yersinia enterocolitica in the environment. The organism is capable of growth in foods and water at low temperatures (0-10°C) (Bottone, 1981; Highsmith et al., 1977). Dominowska and Malottke (1971) found that Y. enterocolitica survived 38 days in the spring and 7 days in summer when kept outdoors in surface waters. Current evidence suggests that Y. enterocolitica may survive for long periods of time in cool, clean waters with a minimum of bacterial competition (Feachem et al., 1983).

TABLE 6-3

t_{90} Values in Hours for Various Types of *V. cholerae* in Various Waters and Wastewaters*

Type of Water Environment	Classical 01			El Tor 01		
	No.	Arithmetic Mean	Range	No.	Arithmetic Mean	Range
Dechlorinated tap water	8	22	3-48	8	49	2-163
Wellwater	1	36	NA	13	116	5-264
Surface water	8	18	0.16-36	10	53	1-230
Seawater	3	95	0.36-161	7	56	3-235
Sewage	1	12	NA	9	66	8-240
Sterilized well-water, surface water or sewage	7	34	3-65	9	59	32-168

*Source: Feachem et al., 1983

No. = Number of results

NA = Not applicable

Little information is available on the survival of Campylobacter jejuni, and no information is available on its survival in domestic sludges or soil. Blaser et al. (1980) found a 7-log reduction in autoclaved streamwater took 5-33 days at 4°C and 2-4 days at 25°C.

Information on Leptospira survival in sludges appears to be non-existent. Leptospira organisms are rapidly inactivated under anaerobic conditions and are very sensitive to inactivation at temperatures >40°C (Feachem et al., 1983). They survive best in soil under high moisture conditions at near neutral pH. In marshy areas where the moisture content was 40-60% and the pH 6.9-7.4, Karaseva et al. (1973) found survival was from 4-15 days. Spinu et al. (1963) reported that leptospires survived for 2-5 days in streamwater at 22-26°C. Diesch (1971) recorded a 3-day survival period in streamwater and wellwater. Animal slurries and sludges are more likely to contain Leptospira. Diesch (1971) found leptospires able to survive 61 days in an oxidation ditch receiving cattle manure. In contrast, survival was only 4 days in sludge from a cattle manure-settling chamber. Lower survival in the sludge was believed due to the absence of oxygen.

6.1.3. Protozoa. Many of the same factors affecting enteric virus and bacteria survival also affect protozoa survival (for example, pH, moisture, temperature and organic content).

Cysts are more susceptible to adverse environmental effects, such as drying and elevated temperatures, than are the eggs of helminths (Kowal, 1985). Several researchers observed the survival times of Entamoeba histolytica cysts in water solutions to be as follows: 4 days (Beaver and Deschamps, 1949); 6-7 days at 10°C, 3 days at 30°C and 1 day at 40°C (Chang and Fair, 1941); and 153 days at 12-22°C (Boeck, 1921). No information is available on survival in sludge-soil mixtures, but survival may be expected

to be similar to survival in water. Coccidian oocysts can remain viable in soil for 15 months (Griffiths, 1978), but E. histolytica cysts died within 5 minutes after drying. Under agricultural field conditions, they survived 42 hours when the soil was wet, and 18 hours when the soil was dry (Rudolfs et al., 1951b). Under optimum conditions of temperature (28-34°C) and moisture, E. histolytica cysts survived at least 8 days in the soil. The optimum soil for cyst survival was found to be dark loam containing 30-50% sand. Soil samples with high proportions of either clay or coarse sand resulted in the lowest cyst survival times (Beaver and Deschamps, 1949). The cysts will die rapidly if dried or frozen.

Giardia lamblia cysts survive in water for 32 days at 10-22°C (Boeck, 1921). Using excystation to determine viability, Giardia survival in tap water was found to be 6 days at 37°C, 25 days at 21°C and 77 days at 8°C (Bingham et al., 1979).

Cryptosporidium sp., which is now known to be present in sewage (Musial, 1985), appears equally resistant as Giardia cysts to chlorine disinfection (Angus, 1983) and may survive prolonged periods at low temperatures (Anderson, 1985).

6.1.4. Helminths. The general consensus is that Ascaris eggs are the most resistant of all the enteric pathogens to adverse environmental conditions (Cram, 1943; Jackson et al., 1977; Meyer et al., 1978). Several researchers have observed extended survival times of Ascaris eggs in soils: Griffiths (1978) found a 4-year survival time and Jackson et al. (1977) observed at least 3 years. Other researchers found that the eggs survived on a drying bed for 66 days (Wright et al., 1942). Soil moistures of >75% (Rudolfs et al., 1951a) or <20% (Reimers et al., 1980) were lethal to Ascaris eggs. The lowest moisture levels at which all Ascaris eggs were

inactivated was seasonal: 5% in fall, 7% in winter, 8% in spring and 15% in summer (Reimers et al., 1981). Eggs were observed to survive for 60-80 days when the moisture content of the soil was <6%, and the temperature was >40°C (Cram, 1943). Refrigerated Ascaris eggs have survived for ≥20 years (Jackson et al., 1977).

Trichuris eggs may remain viable in soil for 6 years (Griffiths, 1978). Hookworm eggs survived 60-80 days with soil conditions of 6% moisture and temperature >40°C (Cram, 1943). At 45°C hookworm larvae survive <1 hour; at 0°C <2 weeks; and at -11°C <24 hours. Hookworms survive best in shaded sandy or loam soils covered by vegetation and protected from drying and excessive wetness. Clay soil, which packs tightly, is unsuitable for survival (Metro, 1983).

One investigation studied the survival of Taenia saginata eggs in sewage, water, liquid manure and on grass. The survival times were 16, 33, 71 and 159 days, respectively (Metro, 1983).

Toxocara eggs were inactivated when the moisture content of the soil was 20% (Smith et al., 1980). Another study observed that moisture and temperature were responsible for inactivation of Toxocara eggs. The lowest moisture levels at which all Toxocara eggs were inactivated were 5% in the fall, 7% in the winter, 8% in the spring and 15% in the summer (Reimers et al., 1981).

The U.S. EPA sponsored a study on the presence of parasites in land-applied sludges at 12 sites nationwide (Theis et al., 1978). The soils were tested only at sites that had received sludge applications for a minimum of 5 years. In Springfield, MO, 50% of the sludge samples and 13% of the soil samples contained parasites. Toxocara was the only parasite found in the soil, while Toxocara, and to a lesser extent Ascaris, were found in the sludge. In Hopkinsville, KY, the soil samples were negative,

while 50% of the sludge samples contained Toxocara as well as some Ascaris. In Frankfort, IN, the soil samples were negative, while 87.5% of the sludge samples were positive with Ascaris, Toxocara, Trichuris and hookworm. In Macon, GA, 7.6% of the 13 soil samples tested were positive for Ascaris only. In sludge and soil samples from Kendalville, IN, Columbus, IN, Wilmington, OH, and Chippewa Falls, WI, no parasites were recovered (Theis et al., 1978).

Anaerobically digested sludge from Oakland, CA, was sprayed onto irrigated crop test plots and onto dryland pasture. The application rates ranged from 7.4-72.4 dry metric tons/hectare. Throughout a 2-year period soil samples from lower application rate areas were positive for parasites in 12 out of 120 samples, and in 21 out of 124 samples from higher application rate areas. The control plot, where no sludge was directly applied, was positive for parasites in 7 out of 75 samples. This indicates either a high endemic parasite population, contamination from the test plots or a combination of both. The parasites found in order of frequency were: Ascaris, Toxascaris, Toxocara and Strongyloides (Theis et al., 1978).

6.2. SUMMARY OF FACTORS CONTROLLING MICROBIAL SURVIVAL

The major factors that influence microbial survival in the environment are listed in Table 6-4. In sewage sludge pH, temperature and moisture are the most important factors in controlling the survival of pathogens. Moisture content of the sludge or sludge-soil mixtures would be greatest in moist soil and during periods of high rainfall. The type of soil is also critical in regard to moisture content; survival is less in sandy soils with greater water-holding capacity, such as loam and muck. Acidic conditions in soil or water can greatly increase bacterial die-off rates (Gerba et al., 1975). While more resistant to inactivation under acidic conditions, both viruses and parasites are inactivated at extremes in pH. The presence of

TABLE 6-4

Factors That Influence the Survival of Enteric Pathogens in the Environment*

Parameters	Survival of Pathogens
Temperature	Increased survival at lower temperatures
pH	Shorter survival at extremes
Desiccation and soil moisture	Increased survival in moist soils
Organic matter	Increased survival and possible growth of bacteria
Antagonism from soil microflora	Increased survival time of bacteria in sterile soil; no clear trend for viruses
Type and strain of organism	Survival dependent on both type and strain

*Source: Gerba et al., 1975

antagonistic microbes, such as protozoa, has a detrimental effect on bacterial survival in soil. The role of biological antagonism against viruses and protozoa in the soil-sludge environment is currently unclear and its significance remains to be determined. Survival times of all enteric pathogens are increased at lower temperatures. While freezing temperatures may kill bacteria and protozoa, they have little effect on viruses and actually increase their survival. Low nutrient availability decreases bacterial survival. In the case of sludges it appears that significant nutrients are available to greatly prolong the survival of indicator bacteria (Donnelly and Scarpino, 1984). The presence of organic matter increases the survival of enteric viruses, and the adsorption or association of viruses with soil or sludge particles also extends their survival time (Hurst et al., 1980a).

The previous review suggests that the order of persistence in the environment, from the longest to shortest survival time, is as follows: helminth eggs < viruses < bacteria < protozoan cysts. In the case of indicator bacteria such as coliforms and fecal coliforms, regrowth may occur in buried sludges (Donnelly and Scarpino, 1984).

To determine risks associated with the landfilling of sludge, it is necessary to be able to predict pathogen survival. Of all the factors known to influence pathogen survival, temperature is the most useful in predicting survival times. The next section is a review of attempts to develop models for predicting viral and bacterial decay in water and soil. Insufficient information is available at present for the development of models for predicting survival of helminths and protozoan cysts.

6.3. MODELS FOR PREDICTING MICROBIAL DIE-OFF IN THE ENVIRONMENT

6.3.1. Viruses. Virus inactivation in water and soil has usually been described as a first-order reaction (Hurst et al., 1980a; Reddy et al., 1981; Vilker, 1981; Yates et al., 1985). Nonlinear survival curves may result if viral aggregates are present or a significant variation exists in sensitivities among the viral population to the factors causing inactivation. The decay rate or inactivation described by a first-order reaction rate expression would be:

$$\text{Decay rate} = \frac{dC}{dt} = -kC$$

where C is infective virus concentration at time t and k is the first-order inactivation constant ($time^{-1}$). Here k would be an expression of the sum total of all factors that influence virus inactivation. Measurement of virus decay has been conducted on a wide variety of surface waters, but such information on soils and groundwater has been limited until recently. Values on virus inactivation were obtained from anaerobic sludge digestion and soil column studies by Reddy et al. (1981). Values used by Reddy et al. (1981) were developed from virus inactivation studies during anaerobic digestion of sludge and from soil columns flooded with sewage. In their model calculations, Matthess and Pekdeger (1985) used values as presented for surface waters by Akin et al. (1971). Grosser (1985) reviewed virus decay observed in a number of environments and used a variety of values for virus decay in his model calculations. Vilker (1981) discussed in detail virus inactivation observed in various environments, but values for groundwater had not been determined at the time. In general, most of the previous work has lacked experimentally determined values for k in groundwater and soil.

Only a few reports exist on decay rates for viruses in groundwater. Keswick et al. (1982a) reported decay rates of 0.19 $\log_{10}$ day^{-1} for coxsackievirus B3 and 0.21 $\log_{10}$ day^{-1} for poliovirus in water from an 84 m deep well with water temperature ranging from 3-15°C in Houston, TX. In wellwater from Florida, Bitton et al. (1983) observed a 0.0456 $\log_{10}$ day^{-1} decay rate for poliovirus 1. In the most extensive study to date, Yates and Gerba (1985) found a mean decay rate of 0.1615 for MS-2 coliphage, poliovirus and echovirus 1 $\log_{10}$ day^{-1} in 11 groundwater samples collected from around the United States. Such information, while providing an idea of decay rates for a particular virus that could be used in the development of a model, does not provide information that can be applied to specific locations since environmental conditions may vary widely. To avoid testing each site in question, information is needed on the relative importance of factors that can be used to predict virus decay. With this in mind, Yates and Gerba (1985) studied the influence of various factors likely to be useful in predicting virus decay in groundwater. They found that groundwater temperature was the single most important predictor of virus decay. Linear regression analysis gave a correlation coefficient of 0.88, which was significant at the 0.01 level. The coefficient of determination was 0.775, meaning that 77.5% of the variation in decay rates among samples could be explained by temperature. The decay rate for coliphage MS-2 as a function of temperature was expressed by the following equation:

Decay rate = ($\log_{10}$ day^{-1}) = -0.181 + 0.0214 x temperature (°C)

Viruses persisted for longer periods of time in wellwater samples than have been found in experiments using surface waters incubated at similar temperatures (Yates et al., 1985).

It is also important to note that examination of the equation developed by Yates et al. (1985) indicates that as groundwater temperatures approach ~8°C decay becomes negligible (Figure 6-1). It is probable that virus decay occurs at these temperatures but over much greater periods of time than could be observed in laboratory experiments covering a period of 3-4 months.

In Canada, the Ontario Ministry of the Environment (Metro, 1983) determined the survival times of parasites from farmland application of sludge. Large numbers of Ascaris, Toxocara and Taenia eggs were then mixed with sewage sludge and applied to the surface of grass and bare soil, and then mixed with the top layer of soil. Conditions were monitored and samples taken periodically to determine the state and number of remaining eggs. It was concluded that on well-drained, bare soil exposed to full sunlight, Ascaris eggs would not survive a full year. When the sludge was mixed with the soil, Ascaris eggs were not recovered below 2 cm after 15 days, suggesting a shorter survival of parasite eggs when this method is used.

Some Ascaris eggs in sludge could be expected to survive for several years, although the numbers would decrease with time. Toxocara eggs had similar survival times to Ascaris.

Strongyloides stercoralis exists in sewage as a delicate larva and does not survive most sewage treatment processes. S. stercoralis larvae typically live for <3 weeks, even under optimal conditions (Feachem et al., 1983). The optimal conditions for the infective filariform larvae are 20-25°C and high moisture. Free-living larvae may actually penetrate to depths of at least 30 cm in soil (Shablovskaya, 1963).

6.3.2. Bacteria. Reddy et al. (1981) attempted to define microbial die-off in water and soil assuming first-order kinetics. First-order die-off rate constants (k) were calculated from a review of the literature. The

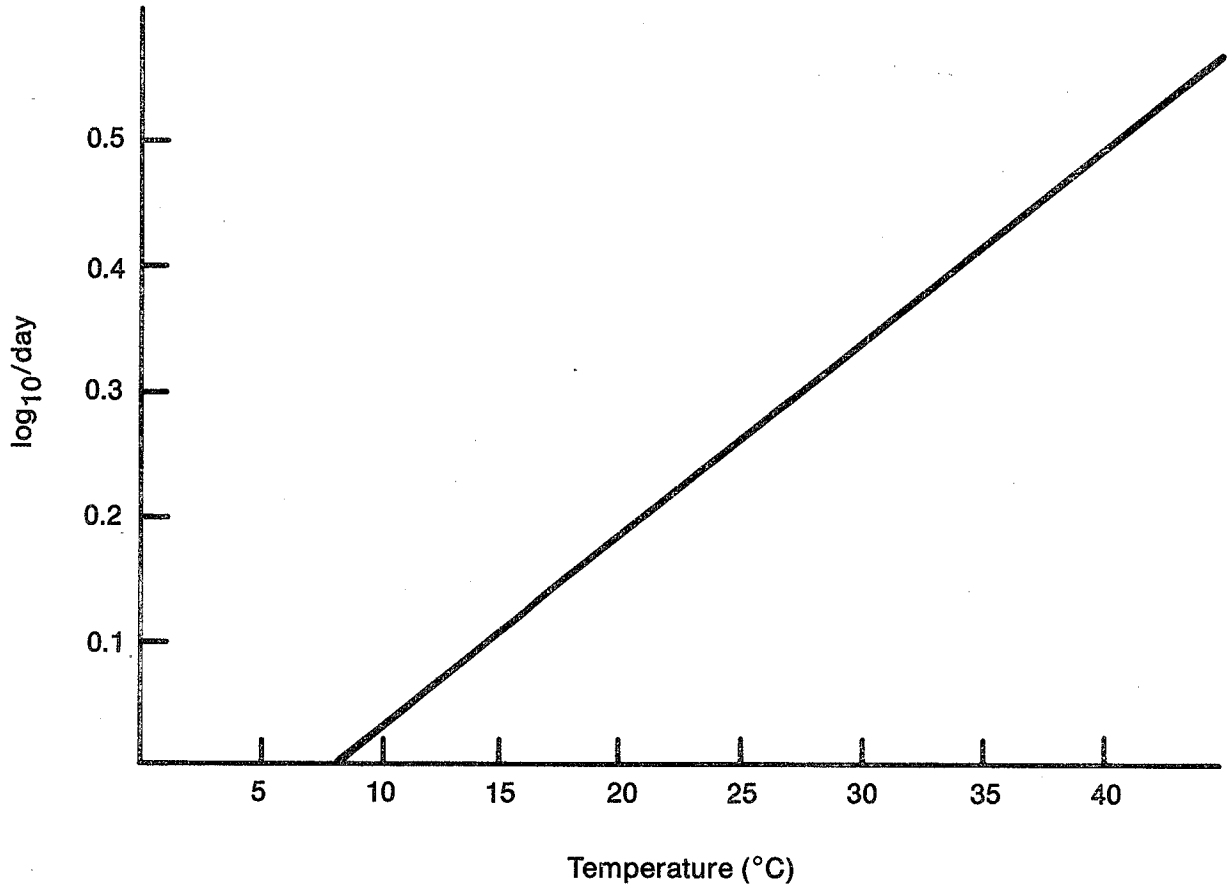


FIGURE 6-1

Virus Decay Rate as a Function of Temperature

Source: Yates, 1985

average die-off constants (day^{-1}) for indicator organisms, Salmonella spp. and Shigella spp. are summarized in Table 6-5. The minimum and maximum die-off rate values covered a large range because studies were conducted under a wide variety of environmental conditions.

6.4. ASSESSMENT OF PATHOGEN SURVIVAL AT SLUDGE LANDFILLS

The relationships between pH, temperature and moisture are the most important and have been quantified with regard to their impact on pathogen die-off. For landfills temperature is probably the most significant factor in predicting pathogen die-off. Since sludge at most landfills is covered the same day it is disposed, moisture losses are likely to be minimal. Even in arid regions drying would be greatly retarded after burial. The pH of raw primary sludge ranges from 5-8 (typical pH 6); anaerobically digested sludges have pH ranges from 6.7-7.5 (typical pH 7.0), and for aerobically digested sludges pH ranges are 5.9-7.7 (typical 7.0) (U.S. EPA, 1974, 1978). Most enteric pathogens are very stable in these pH ranges, and pH would not have a major effect on their survival. Only in lime-conditioned sludges where the pH may be >10.0 will pH have a significant impact on pathogen survival. At pH levels $\geq$10 greater pathogen survival is substantially decreased.

When significant amounts of organic matter are present, the survival of indicator bacteria, such as total coliforms, is prolonged by years. (Donnelly and Scarpino, 1984). Insufficient information is available to predict survival of Salmonella or other enteric bacterial pathogens in buried sludge, although it is probably less than that of the indicator bacteria. Helminth eggs will probably survive for several years. Viruses could also survive for prolonged periods in buried sludge. Since temperature is a dominant factor in virus survival, it should be possible to

TABLE 6-5

Average Die-Off Rate Constants (day^{-1}) for
Selected Microorganisms in a Soil-Water-Plant System[a]

Microorganisms	Average		Maximum		Minimum		S.D.	Observations
	K[b]	Half-life (hours)	K[b]	Half-life (hours)	K[b]	Half-life (hours)		
Escherichia coli	0.92	18.1	6.39	2.6	0.15	110.9	0.64	26
Fecal coliforms	1.53	10.9	9.10	1.8	0.07	237.6	4.35	46
Fecal streptococci	0.37	45.0	3.87	4.3	0.05	332.7	0.69	34
Salmonella spp.	1.33	12.5	6.93	2.4	0.21	79.2	1.70	16
Shigella spp.	0.68	24.5	0.74[c]	22.5	0.62[c]	26.8	0.06	3

[a]Source: Reddy et al., 1981

[b]Die-off rate constant (k); $M_X = M_0 \exp(-kT)$ where M_X = final concentration, M_0 = initial concentration, T = time (days).

[c]The maximum and minimum die-off rate constants were apparently reversed in Reddy et al. (1981) and have been corrected here.

S.D. = Standard deviation

estimate virus survival with a predictive model. Sufficient data appear to be available to estimate viral and bacterial decay in groundwater and perhaps soil. However, insufficient data are available on viral and bacterial decay in sludges at different temperatures to be used in a predictive model. Protozoan cysts probably survive for a shorter period of time than the other pathogens, but survival data in sludges are lacking.

In summary, it appears that if sufficient information were available, predictive models for pathogen decay in sludge landfills, soil and groundwater could be developed. Survival times could be predicted on the basis of sludge-soil type, pH, temperature and moisture.

7. TRANSPORT OF PATHOGENS IN THE SUBSURFACE

In conjunction with the survival rates, knowledge of pathogen movement through the sludge-soil matrix is critical. Factors affecting bacterial movement in soil include physical characteristics of the soil, such as texture and pore size, as well as environmental and chemical factors, such as temperature and dissolved salts. For example, retention by soil particles is great for soils with a high clay content, and movement of the pathogens through the soil profile is substantially reduced. Therefore, groundwater contamination would not be a major route of exposure with clay soil conditions unless cracks or fissures are present. By contrast, sand and gravel permit greater and more rapid movement (Table 7-1). Major factors which determine the extent of microbial movement are shown in Table 7-2. Of these, size of the microorganisms is probably the most important. In most soils viruses could be expected to travel the greatest distance because of their small size, while the movement of protozoa and helminths would be more limited (Table 7-3) because of their large size.

7.1. VIRUSES

Virus movement in groundwater has been demonstrated under both laboratory and field conditions. Keswick and Gerba (1980) reviewed instances of virus isolation from groundwater. The published data indicate that viruses can travel at least 67 m vertically and 408 m laterally in soil. In gravel and karst substrata, viruses have been observed to travel as far as 1600 m at rates as high as 100 m/hour (Keswick et al., 1982b).

The major factors that affect virus migration in the subsurface are listed in Tables 7-2 and 7-4. The major mechanism of virus removal in soil is by adsorption to soil particles (Gerba and Bitton, 1984). This is in

TABLE 7-1

Hydraulic Conductivities of Subsurface Material*

Saturated Granular Material	Hydraulic Conductivity (cm/day)
Clay soils (surface)	0.01-0.2
Deep clay beds	10^{-8}-10^{-2}
Loam soils (surface)	0.1-1
Fine sand	1-5
Medium sand	5-20
Coarse sand	20-100
Gravel	100-1000
Sand and gravel mixes	5-100
Clay, sand and gravel mixes (till)	0.001-0.1

*Source: Adapted from Bouwer, 1984

TABLE 7-2

Soil Factors Affecting Infiltration and Movement of Microorganisms in Soil*

I.	Physical characteristics of soil
	a. Texture
	b. Particle size distribution
	c. Clay type and content
	d. Organic matter type and content
	e. pH
	f. Cation exchange capacity (CEC)
	g. Pore size distribution
II.	Environmental and chemical factors of soil
	a. Temperature
	b. Moisture content
	c. Soil water flux (saturated vs. unsaturated flow)
	d. Chemical make-up of ions in the soil solution and their concentrations
	e. Microbial density and dimensions
	f. Nature of organic matter in waste effluent solution concentration and size

*Source: Modified from Crane and Moore, 1984

TABLE 7-3

Sizes of Waterborne Bacteria, Viruses and Parasites*

Microorganism	Size (μm)
Bacteria	1-10
Salmonella typhi	
Shigella dysenteriae	
Escherichia coli	
Vibrio cholerae	
Viruses	0.02-0.08
Enteroviruses (polio, echo, coxsackie)	
Rotavirus	
Norwalk-like virus	
Hepatitis A	
Adenovirus	
Protozoa (cysts)	5-20
Giardia lamblia	
Entamoeba histolytica	
Helminths (eggs)	25-38
Ascaris spp.	
Taenia spp.	
Fungi (spores)	35-40
Aspergillus spp.	

*Source: Bitton and Gerba, 1984

TABLE 7-4

Factors Affecting Virus Transport in Soil*

Factor	Comments
Soil type	Fine-textured soils retain viruses more effectively than light-textured soils. Iron oxides increase the adsorptive capacity of soils. Muck soils are generally poor adsorbents.
pH	Generally, adsorption increases when pH decreases. However, the reported trends are not clear-cut due to complicating factors.
Cations	Adsorption increases in the presence of cations (cations help reduce repulsive forces on both virus and soil particles). Rainwater may desorb viruses from soil due to its low conductivity.
Soluble organics	Organisms can compete with viruses for adsorption sites. Humic and fulvic acid reduce adsorption to soils.
Virus type	Adsorption to soils varies with virus type and strain. Viruses may have different isoelectric points.
Flow rate	The higher the flow rate, the lower virus adsorption to soils.
Saturated vs. unsaturated flow	Virus movement is less under unsaturated flow conditions.

*Source: Bitton and Gerba, 1984

contrast to bacteria, protozoa and helminths, which are primarily removed by filtration and straining. Microbial adsorption to soil particles is believed to be mediated by a combination of electrostatic and hydrophobic interactions (Gerba, 1984b). These interactions are influenced by the factors listed in Table 7-2. For example, viruses adsorb more readily to clayey soils than to sandy soils, and adsorption is enhanced in the presence of divalent cations (Gerba, 1984b). In gravel substrata, no virus adsorption may occur (Grondin and Gerba, 1986).

Once a virus is adsorbed to a soil or sludge particle, it is not necessarily permanently immobilized. A reduction in the ionic strength of the water content in the soil, which can be induced by rainfall, can cause viruses to desorb and migrate further in the subsurface (Lance et al., 1976; Sobsey, 1983). This phenomenon was observed in a field study by Wellings et al. (1974), when wells at a wastewater land application site in Florida, which had previously been virus free, were found to contain viruses after a period of heavy rainfall. Eluted viruses occurred as a burst.

Adsorption of the virus to the soil and sludge also appears to be highly dependent upon its isoelectric point (Gerba et al., 1979, 1982). Thus, some viruses such as poliovirus adsorb more readily to soils and are less likely to be eluted than others (Goyal and Gerba, 1979; Zerda et al., 1985; Landry et al., 1980) (see Table 7-4). In recent studies, Sobsey (1985) showed that hepatitis virus adsorbs significantly less to sandy soils than poliovirus type 1. Its adsorptive behavior appears to more closely resemble that of echovirus type 1. In some sandy soils bacteria may actually be removed less effectively than poliovirus because of virus adsorption to the soil (Wang et al., 1985).

7.1.1. Land Application. Virus binding to the sludge is also significantly influenced by pH. Ait et al. (1984) found that viruses bind well to sludges at pH 5-7, but above pH 7.0 binding decreases rapidly. Little virus adsorption occurred between pH 8-9 (Table 7-5). Thus, viruses may be more mobile when high pH sludges are disposed.

Results of previous research indicate that viruses are tightly bound to sewage sludges and not easily released (Bitton et al., 1978; Farrah et al., 1981; Damgaard-Larson et al., 1977; MSDGC, 1979) (Table 7-6). Damgaard-Larson et al. (1977) and Farrah et al. (1981) could not detect any virus movement from surface-applied sludge. Bitton et al. (1978) only observed movement when freshly applied liquid (nonair-dried) sludges were applied. Only 0.2% of the poliovirus type 1 was observed in a percolate of a 54-cm column. Moore et al. (1977) observed poliovirus type 1 movement through 7 cm of soil when air-dried anaerobically digested sludge was applied. Between 0.2-2% of the virus that was applied was found in the soil percolate.

In contrast to previous studies, Jorgensen and Lund (1985) were able to isolate naturally occurring enteroviruses from a 3 m deep well at a site where anaerobically digested sludge was applied to diluvial sand in a forest plantation. The 30 ℓ sample contained both poliovirus type 2 and coxsackievirus B3. It was collected the 11th week after sludge application. In laboratory studies, Jorgensen (1985) observed elution and rapid movement of coxsackievirus B3 through 100 cm of sandy loam soil after application of anaerobically digested sludge under saturated flow conditions. In contrast, no viruses were detected in percolates from columns of sandy soil. At the end of the experiment, viruses were detected in soil eluates at depths $\leq$30 cm in the sandy loam soil, but viruses could be detected

TABLE 7-5

Effect of pH on Poliovirus Adsorption to Sewage Sludge*

pH	Percent of Bound Poliovirus
5.0	42
6.0	42
7.0	42
7.5	35
8.0	28
9.0	10

*Source: Adapted from Ait et al., 1984

TABLE 7-6

Summary of Virus Migration from Sludge Through Soils

Soil Type	Type of Sludge	pH of Soil Percolate or Soil	Distance of Migration (cm)	Percent Virus Eluted	Reference
Sand	anaerobic-dewatered	6.0	125	0	Damgaard-Larson et al., 1977
Sandy loam	anaerobic-2% solids-air-dried	4.5-6.0	54	0.2	Bitton et al., 1978
Sand	anaerobic	3.5-4.0 3.5-4.0	14	0 0	Jorgensen, 1985
Sandy loam	anaerobic	7.0-8.0	100	1.0	Jorgensen, 1985
NR	anaerobic	NR	7	0.2-2	Moore et al., 1977

NR = Not reported

only to a depth of 14 cm in the column containing sand. In contrast virus movement was <3.5 cm when the same columns were run under aerobic, unsaturated flow conditions.

A review of the studies on virus movement from land-applied sludges suggests two major reasons for the elution and penetration of viruses through soil observed by Jorgensen (1985). As shown in Table 7-6, the studies prior to Jorgensen used acid soils, which would be expected to tightly bind any adsorbed virus. Secondly, these studies were conducted with unsaturated soil. It should be pointed out that both poliovirus type 1 and coxsackievirus B3 adsorb to a much greater degree to sludge and soils than many of the other enteroviruses (Gerba et al., 1979) and, thus, greater movement of other viruses would be expected to occur. Also, in the previous studies sludge was applied to the soil surface and not buried as occurs in landfills. Virus inactivation would be expected to be greater in sludge applied on the soil surface.

7.1.2. Transport. Recently, Yates et al. (1985) reviewed various proposed models for virus transport in the subsurface. Two basic approaches have been used to model virus transport: one relies on the assumption of instantaneous equilibrium between the suspended and adsorbed virus concentrations (Grosser, 1985) and the other uses a mass-transfer or "rate-controlled" model to account for the distribution of viruses between the fluid and solid phases (Vilker and Burge, 1980). In both cases, after several assumptions have been made, the model formulation results in linear partial differential equations, which can be solved by a variety of methods (either analytically or numerically) depending upon the problem domain, heterogeneities in aquifer properties and boundary conditions. All of the models are based on solute transport; however, viruses are colloids and may

behave differently than solutes during transport through an aquifer. For example, Grondin and Gerba (1986) recently found that MS-2 coliphage moves at a velocity 1.5-1.9 times faster than the average groundwater flow through coarse media. At this time, neither approach is more correct than the other, as the models are only approximations of observed laboratory phenomena. In both cases, the mathematical capabilities far exceed what is currently known about the behavior of viruses in soils and groundwater. Laboratory experimentation and field verification with different substrata are needed to validate the usefulness of these models. However, they could be used as a first approximation based on existing data.

Using actual field data on virus decay, Yates et al. (1985) employed kriging, a geostatistical technique, to estimate separation distances between drinking-water wells and sources of contamination. No virus adsorption was assumed in the model. A safe distance was defined as the time of travel (regional groundwater flow) that would result in 7 logs of virus inactivation. A 7-log reduction was chosen since this would reduce the assumed average enteric virus concentration below detection in the groundwater (that is, <1 virus/1000 ℓ). An even more simplified approach is that of Wang et al. (1981) who found that virus removal through sandy soils could be predicted by the flow rate. Thus, at a flow rate of 33 cm/day the rate of poliovirus removal was 0.04 log/cm, while at 1352 cm/day it was 0.007 log/cm. It is important to note that most of the observed virus removal occurred near the soil surface and was significantly less in the lower depths of the column (Table 7-7).

Almost no studies on virus movement in the subsurface have been made under conditions of unsaturated flow. This is because of the difficulty in obtaining water samples under unsaturated flow conditions. In studying poliovirus type 1 movement through loamy sand, Lance and Gerba (1984) found

TABLE 7-7

Rate of Virus Removal Through Different Soil Types*

Soil Type	Virus	Flow Rate cm/day	Rate of Removal (log_{10}/cm) Column Depth	
			0-17 cm	17-87 cm
Rubicon sand	Polio 1	314	0.028	0.005
	Echo 1	282	0.032	0.003
Anthony sandy loam	Polio 1	33	0.144	0.02
FM loamy sand	Polio 1	75	0.088	0.015
	Echo 1	76	0.046	0.025
	f2	75	0	0
Gravel	f2	75	0	0
	MS-2	75	0	0

*Source: Wang et al., 1981

that poliovirus movement was substantially less under unsaturated flow conditions. Viruses did not move below the 40-cm level when sewage water was applied at less than saturation. However, field studies at sites where wastewater irrigation of food crops is practiced (Goyal et al., 1984) suggest that enteroviruses can travel at least several meters through the unsaturated zone.

While laboratory studies are essential to an understanding of the basic mechanisms of microbial fate and transport in soils and groundwater, ultimate validation of actual behavior rests with field observations. Without such validation critical errors could be made in assessing microbial behavior. Apparently no studies have been conducted to determine if viruses gain entrance into the groundwater from sludge landfills. Gerba (1986b) recently reviewed field studies on the transport and fate of viruses from septic tanks and sites where land application of wastewater is practiced. The results suggest that naturally occurring enteroviruses can travel substantial distances. Table 7-8 shows the removal of enteroviruses observed at several field sites where land application of wastewater is practiced. The observed virus removal ranged from 0.023-0.49 log/m. All were sandy to gravel soils. Stramer (1984) dosed several septic tank systems with single doses of poliovirus type 1 vaccine derived from cell culture or from stools of recently vaccinated infants. The viruses were demonstrated to persist for several months in each of the septic tank systems, and groundwater contamination was demonstrated at all sites studied. At one site viruses passed from the septic tank and traveled 50 m through silt loam soil and were detected in water from a nearby lake 43-109 days after dosing. Little virus removal occurred during transport through the unsaturated zone.

TABLE 7-8

Observed Enterovirus Removals During Field Studies

Site	Purpose	Soil Type	Observed Virus Movement (m)		Depth of Unsaturated Zone (m)	Enterovirus (PFU/ℓ)		Removal (log/m)	References
			Vertical	Horizontal		Applied	Observed in Groundwater		
Lubbock, TX	irrigation	sandy loam	27.5	NA	3	9.5	0.0025 -0.068	0.13 0.08	Goyal et al., 1984
St. Petersburg, FL	irrigation	sand	3 6	NA NA	ND ND	60 60	2/1 0.17/1	0.49 0.49	Wellings et al., 1974
Phoenix, AZ	rapid infiltration	sand and gravel	30	5	NA	84	0.002	0.13	Gerba, 1986a
Wisconsin	septic tank	silt loam	46	0.9	0.9	501	933	0.006	Stramer, 1984
Parkland, NY	rapid infiltration	coarse sand and fine gravel	46	6	5.5	76	5	0.023	Vaughn et al., 1978
Meadowbrook, NY	rapid infiltration with 1-2% silt	coarse sand and fine gravel silt	3	11	9	62	2	0.11	Vaughn et al., 1978

NA = Not applicable; ND = no data

A comparison of field and laboratory studies suggests that laboratory studies overestimate virus removal. For example, at the Phoenix, AZ, land application site, laboratory studies suggested that at least 2 logs of poliovirus 1 removal could be expected during movement of the effluent through 1 m of soil (Lance et al., 1976). However, field observations suggested removals of only 0.08-0.11 log removal/m. Reasons for lower virus in the field may be due to numerous factors including: (1) naturally occurring viruses are not as easily retained by soils as laboratory strains, possibly because the procedures used to purify the laboratory strains may affect the ability of the virus to be retained by the soil; (2) soils are not homogenous in the field and viruses may be expected to move at different rates throughout a field site; and (3) rainfall and other events in the field have major impacts on virus movement under field conditions.

7.2. BACTERIA

In contrast to viruses, bacterial removal by soil is believed to largely involve filtration, although adsorption also plays a role. Bacterial movement through the soil surface appears to be more restricted than that of viruses, although under the proper conditions bacteria may actually travel greater distances (Wang et al., 1985). Although bacterial movement appears to be limited to depths of 10-50 cm in most soils, travel distances of 3-122 m have been observed in sandy soils, and distances of travel as great as 920 m have been observed through gravel (Crane and Moore, 1984; Lewis et al., 1980).

A review of the data obtained from septic tank studies suggests that in permeable soils (but not in coarse sands or gravels) 1-2 m is adequate for essentially complete bacterial removal (Lewis et al., 1980; Hagedorn, 1980), provided the soil has both a layer permeable to effluent flow and another

region adequately restrictive to form a clogged zone. Also, hydraulic loadings of <50 mm/day appear necessary for efficient removal (Lewis et al., 1980). A summary of coliform removal under unsaturated conditions in sandy soils is shown in Table 7-9. It appears that 4-8 logs of coliform bacteria would be removed for each meter of unsaturated zone provided hydraulic loading is <50 mm/day. A review of coliform bacteria removal at sites where land application of wastewater is practiced indicates that significantly less bacteria would be removed (see Table 7-9).

Rainfall can have a major effect on bacterial migration through the unsaturated zone by lowering ionic concentration and increasing infiltration rates (Gerba and Bitton, 1984).

Several surveys have indicated that rainfall and well depth are related to microbial groundwater quality. Studies in the State of Washington indicate that shallow drinking-water wells contain average median coliform values of 8 MPN/100 mℓ with an average depth of 9.4 m (31 ft), while deep wells with an average depth of 153.3 m (503 ft) average 4 MPN/100 mℓ (Hendricks et al., 1979). It was also observed that virtually all bacterial contamination coincided with the periods following heaviest rainfall. Increased levels of bacterial contamination of drinking-wellwater after periods of rain have been noted in several studies (Gerba and Bitton, 1984). It was also noted that while an increase in coliform bacteria appears almost immediately after periods of heavy rainfall in shallow wells, the increase did not occur until 2 weeks later in deeper wells (Loehnert, 1981). Thus, any satisfactory study of wellwater quality should include sampling during periods of highest rainfall. Sandhu et al. (1979) found that basic well design and construction had little effect on the extent

TABLE 7-9

Coliform Bacterial Removal by Soil

Site	Soil Type	Hydraulic Loading (mm/day)	Log_{10} Removal of Bacteria/m	Soil Moisture	Remarks	References
Wisconsin	loamy sand	50	8.7	unsaturated		Ziebell et al., 1975
Wisconsin	sand loam	24	4.5	unsaturated		Bouma et al., 1974
Florida	fine sand	7	5.8	unsaturated	laboratory lysimeter-pig slurry	Dazzo et al., 1972
Denmark	sand	150	1.3-2.7	unsaturated	land application of sewage	Grunnet and Olesen, 1979
			0.2	saturated		
Colorado	fractured rock	7000	0.05	unsaturated		Allen and Morrison, 1973
Arizona	fine loamy sand	250	0.57	unsaturated	land application of sewage	Gilbert et al., 1976
New York	sand	-	1.1	unsaturated	land application of sewage	Aulenbach et al., 1974
New Zealand	alluvial gravels	-	0.17	unsaturated	land application of sewage	Martin and Noonan, 1977

of microbial pollution in their study area. Unfortunately, few studies have attempted to coordinate bacterial sampling with rainfall infiltration.

If bacteria are able to penetrate to the saturated zone, they appear capable of being transmitted significant distances in sandy and gravel soils, although significant reductions may occur with travel distance (Crane and Moore, 1984). Subsurface conditions may markedly affect bacterial transport. For example, both Rahe et al. (1978) and McCoy and Hagedorn (1980) observed that when confining layers are present or other water-restrictive layers are present, the bulk of bacterial transport occurs directly above the water-restrictive layer. Under such conditions Rahe et al. (1978) observed E. coli movement rates of 1500 cm/hour. They observed that once the organisms move into zones of high permeability, they experience little mixing or dilution, but rather are transported through macropores relatively unaffected by the medium through which they are being moved.

Hagedorn (1980) found fecal coliforms from sludge-amended land in groundwater sampling wells 1 m deep. By the fifth week after application, these numbers fell below the detection limit of 20 cells/100 mℓ and remained so for the duration of the sampling program. Donnelly and Scarpino (1984) found that bacterial indicators (coliforms) were capable of leaching from lysimeters containing sewage sludge for at least 13 weeks. Examination of the sludge in the lysimeters after 2 years demonstrated that the coliforms had not died as was indicated in the leachate.

Sedita et al. (1977) monitored three wells in an area where land application of sludge (30 dry tons/acre or 67.2 metric tons/hectare) is applied by spraying or soil incorporation. No bacteria or viruses were observed in

three monitoring wells at the site. The depth to water was not given in the report. Liu (1982) studied the effect of long-term farmland disposal of anaerobically digested sludge on microbiological quality of groundwater using a lysimeter system. It was observed that after 4 years of heavy surface application of sludge, no coliform or fecal coliform bacteria penetrated 1.8 m of loamy sand or silt loam soil exposed to outdoor conditions and natural rainfall.

More recently, Yates (1985) evaluated the literature on viral and bacterial movement through soil. Yates (1985) compared movement observed by soil type and by percolation rate (application rate) in order to develop a rating system for predicting groundwater potential by microorganisms originating from septic tanks.

In order to develop ratings based on soil type, data on the extent of vertical movement of microorganisms in soil were accumulated from a review of the literature. Some of the data were obtained from column studies conducted in the laboratory, others from field studies. These data were plotted to determine the influence of soil type on the distance that a microorganism was observed to travel in that soil (Figure 7-1). Soil type was plotted as a function of decreasing particle size from fracture rock to fine sand, and as a function of increasing clay content from fine sand to clay (Yates, 1985).

The data were analyzed using linear regression, and a correlation ($r = -0.83$) was found to exist between soil type and the $\log_{10}$ (distance) of movement. The relationship can be expressed by Equation 7-1:

$$y = -0.28928x + 1.7967 \qquad (7\text{-}1)$$

where y equals $\log_{10}$ distance of movement and x equals soil type.

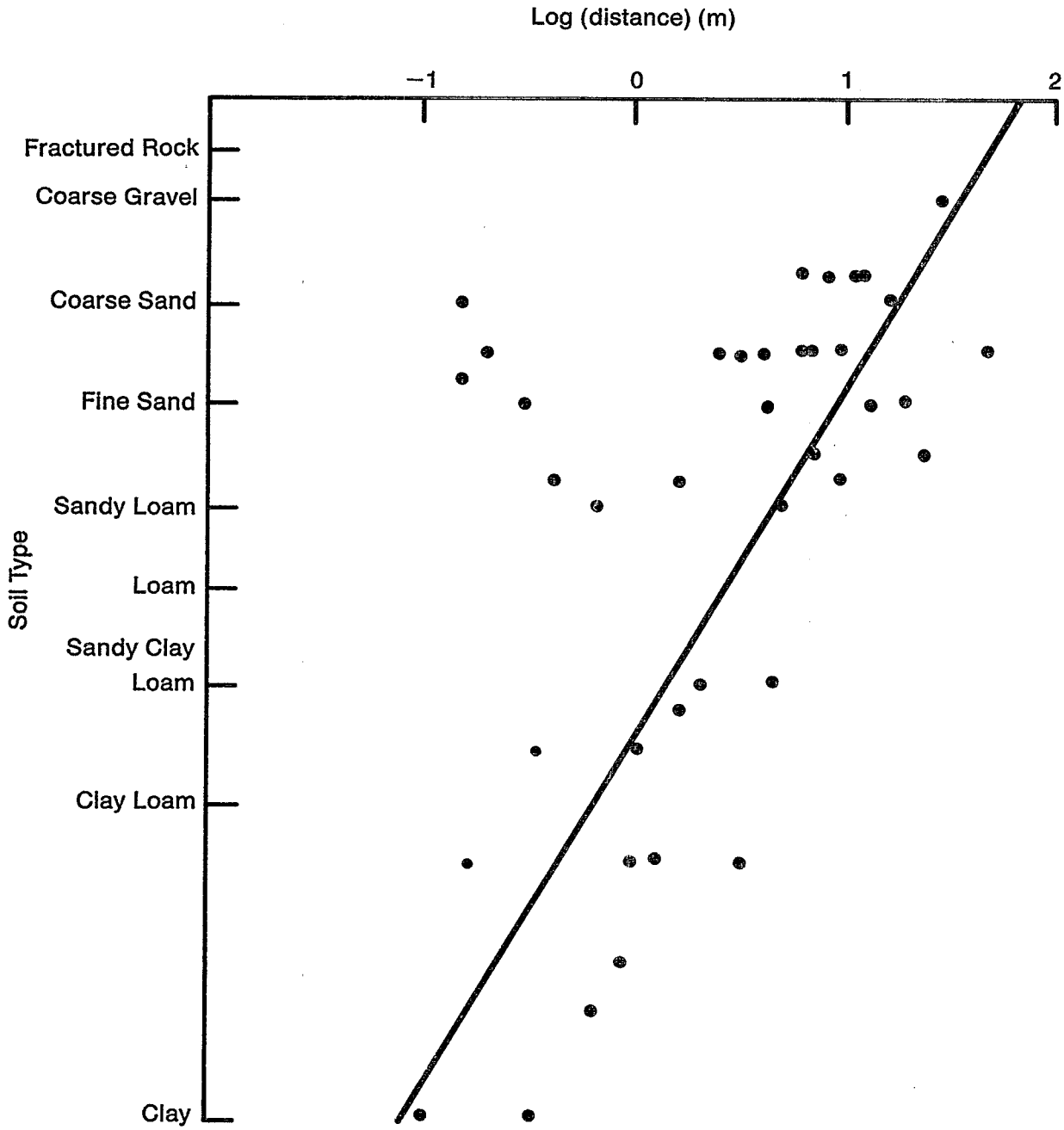


FIGURE 7-1

Vertical Movement of Microorganisms as a Function of Soil Type

Source: Yates, 1985

Once the importance of soil type in limiting microbial movement was verified, ratings had to be developed to reflect this. Yates (1985) felt that soil type in itself was not as important as soil type in relation to the depth to water. In other words, if the site has a shallow water table, and the soil has a clayey texture, the potential for groundwater contamination is much less than if the soil is a coarse gravel. Also, the importance of the depth to water in a clay soil is less than the importance of depth to water in a sandy soil.

Yates (1985) also evaluated the influence of the flow rate of water movement through the soil on removal of bacteria and viruses. Data on the effect of infiltration rate of microbially laden wastewater on the degree of removal of microorganisms in the percolating effluent were obtained by surveying the published literature. The flow rate was found to be correlated ($r = 0.88$) with the degree of removal of microorganisms. This relationship, shown in Figure 7-2, can be expressed by using Equation 7-2:

$$y = -0.53763x - 0.59602 \qquad (7\text{-}2)$$

where y equals $-\log_{10}$ removal per cm and x equals application rate $(\log_{10})$/cm.

7.3. PROTOZOA AND HELMINTHS

Because of their large size the movement of protozoan cysts and helminth eggs would be expected to be even more limited than bacteria. Cram (1943) found no movement of Ascaris eggs, hookworm eggs and Entamoeba histolytica cysts through a 60-cm layer of sand after application of raw settled sludge.

In another study, Taenia saginata eggs were completely removed in a glass cylinder containing a 12-inch (30-cm) column of sand in 3 out of 4 experiments; 99.6% of the eggs were removed in the fourth experiment (Newton et al., 1949).

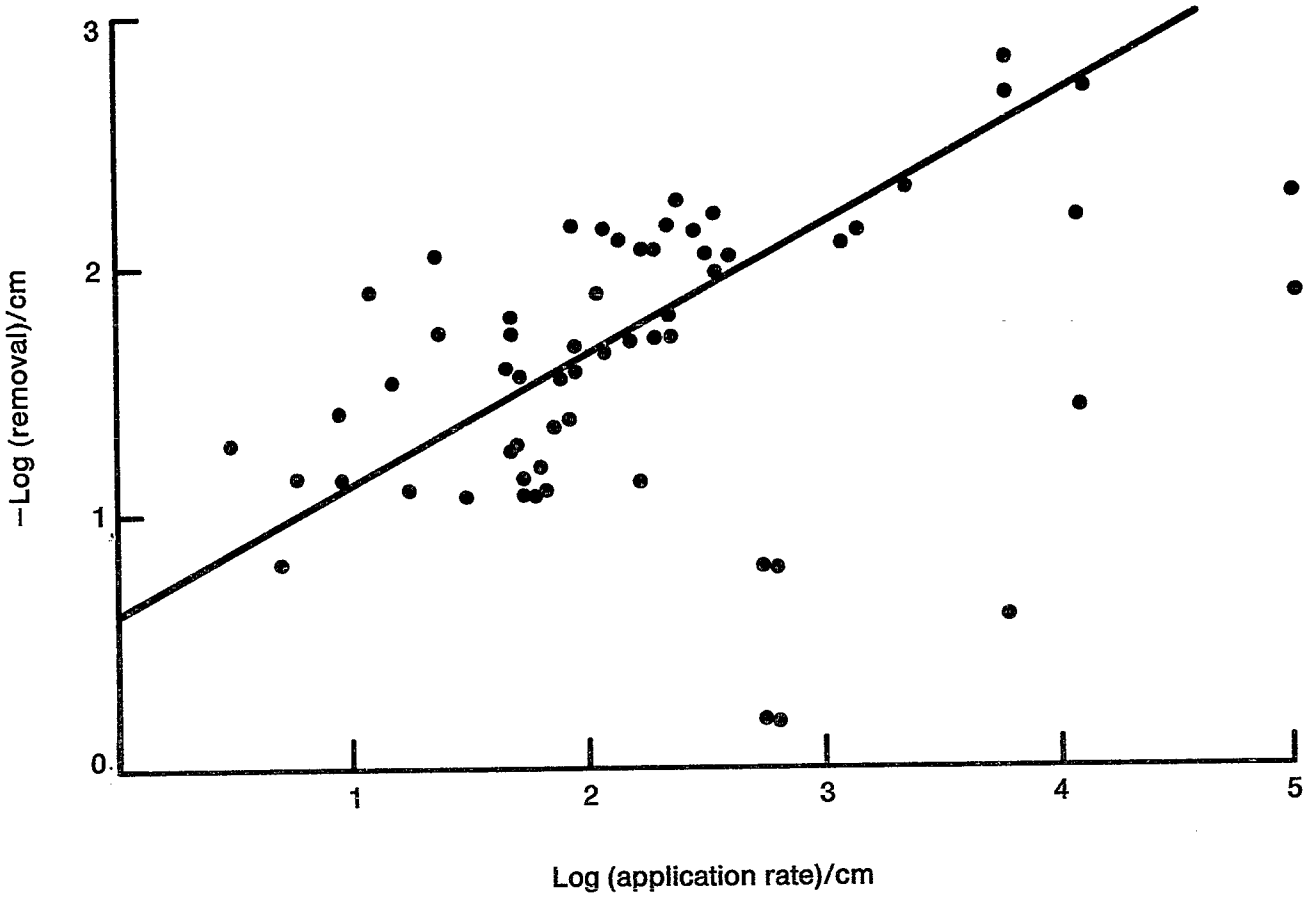


FIGURE 7-2

Removal of Microorganisms as a Function of the Flow Rate of Effluent Through the Soil

Source: Yates, 1985

Digested sludge from Sacramento, CA, was surface spread onto a land disposal site. Testing was conducted over a 2-year period in areas both upstream and downstream of the disposal site. None of the soil or stream samples was positive for parasites (Storm et al., 1979).

In a Canadian soil core experiment using Ascaris-seeded sludge under natural conditions, it was concluded that there was no appreciable downward movement of the parasite eggs, even in well-drained soil. After 15 days, no eggs were recovered below 2 cm. The number of eggs found on grass alone was much lower than when surface soil was included in the sample, indicating that most eggs in the sludge would remain at or near the soil surface (Metro, 1983).

Still, it should be recognized that several outbreaks of waterborne disease attributed to groundwater contaminated by Giardia cysts have occurred in the United States (Jakubowski and Hoff, 1979).

A recent epidemiological study evaluating risk factors associated with endemic giardiasis in the New England area found the use of shallow household wells for drinking water to be a significant risk factor (Chute et al., 1985). Numerous outbreaks of giardiasis have also occurred from surface water that was passed through sand filters. Giardia can penetrate a meter of fine sand (0.28 mm average diameter) (Logsdon et al., 1984). When Giardia cysts were applied to a sand column, 0.1-64% of the cysts were able to penetrate to a depth of 96 cm at operational flow rates of 0.04-0.4 m/hour. No studies could be found on the expected removal of parasites by soils. Ghirose (1986) has reported the isolation of protozoan cysts at several meters below the soil surface. Finally, studies in Russia have shown that some free-living forms of adult Strongyloides stercoralis penetrated to a depth of 0.3 m in soil (Shablovskaya, 1963).

7.4. SUMMARY OF MICROBIAL TRANSPORT THROUGH THE SUBSURFACE

As indicated in this review, many factors influence microbial transport through the subsurface. No information is currently available on the transport of microorganisms in leachate generated from sludge-only landfills. The chemical composition of this leachate would greatly influence transport of microorganisms from the burial site. With all factors considered, viruses have the greatest likelihood of being transported from the site because of their small size. The movement of bacteria, protozoan cysts and helminth eggs would be substantially less. No significant movement of protozoan cysts or helminth eggs would occur unless gravel or fractured substrata was present. Bacterial movement may be limited to a few cm in most soils. However, in sandy soils subject to high rainfall significant movement could occur.

The depth of the unsaturated zone is probably the greatest barrier in preventing microbial movement into the groundwater. However, field studies suggest that viruses may penetrate several meters of unsaturated soil to reach the groundwater (see Table 7-8). Quantitative information on microbial movement through the unsaturated zone is almost nonexistent.

Several models have been developed for predicting microbial transport through the saturated zone (Grosser, 1985; Vilker and Burge, 1980); however, these models have yet to be verified by laboratory and field studies. They are based on solute transport models, which may not be totally useful in predicting microbial transport since microorganisms are colloids (Grondin and Gerba, 1986). These models could be useful as an attempt to approximate microbial movement from sludge landfills.

Through a review of the literature, Yates (1985) has developed a qualitative rating system, which could be used to estimate the probability of microbial transport from a waste site. Again, actual field verification is lacking.

A comparison of field and laboratory studies on virus and bacterial movement in this section suggests that travel of these organisms in the subsurface is greater in the field than laboratory studies would imply. Only through field studies at actual sludge landfills will the real potential for transport be fully understood.

8. EVALUATION OF GROUNDWATER POLLUTION POTENTIAL BY MICROORGANISMS USING MICRO-DRASTIC

A methodology has been described to evaluate the groundwater pollution potential of any site based on its hydrogeologic setting (Aller et al., 1985). A relative rating is given to various factors used to describe the site. These factors include Depth to water table, net Recharge, Aquifer media, Soil media, Topography, Impact of the vadose zone and hydraulic Conductivity of the aquifer. These factors, which form the acronym DRASTIC, are used to infer the potential for contaminants to enter groundwater. The relative ranking scheme uses a combination of weights and ratings to produce a numerical value, called the DRASTIC INDEX, which helps rank areas with respect to groundwater contamination vulnerability. These weights and ratings were determined empirically by a group of experts (Aller et al., 1985).

Using a similar methodology, Yates (1985) developed a rating system to evaluate the potential for groundwater contamination by microorganisms (bacteria and viruses) from septic tanks. Eight factors are used in the rating system: depth to water, net recharge, hydraulic conductivity, temperature, soil type, aquifer medium, application rate and distance to a point of water use. These factors are then ranked in terms of their importance relative to the other factors in influencing the survival and movement of microorganisms through the subsurface. Weights are assigned to each factor, with a weight of 1 signifying the least importance and a weight of 5 signifying the greatest importance. In addition to the weights, which are constant, each factor is assigned a rating based on the conditions found at the particular site being considered. The ratings are determined from

graphs, which have been provided for each factor. The graphs were determined from a review of the literature on microbial survival and transport in groundwater. An index, which gives an indication of the relative potential for groundwater contamination by microorganisms present in septic tank effluent, can then be computed by multiplying each factor rating by its associated weight and summing all the factors.

The factors and weights used in the micro-DRASTIC system developed by Yates (1985) are shown in Table 8-1. The index used to evaluate a site is computed by using Equation 8-1:

$$\text{Index} = 5\,\text{DTW} + 2\text{R} + 3\text{K} + 2\text{T} + 5\text{S} + 3\text{A} + 4\text{AR} + 5\text{D}. \qquad (8\text{-}1)$$

The higher the index, the higher the potential for microorganisms to survive and be transported to the underlying groundwater. The index may range from 0-290 and provides a relative indication of the potential for groundwater contamination by microorganisms. A site with a higher index is more likely to have contamination problems than one with a lower rating. For a more definitive interpretation of the index, Yates (1985) suggested the following scale as a guide:

0-75	not very probable
75-150	possible
150-225	probable
>225	very probable.

Although the system was developed to evaluate the groundwater pollution from septic tanks, it could also potentially be used to evaluate sites that practice land application of wastewater and sludge landfills. To evaluate landfills the factor for application rate would not be used since effluent is not being applied. Potentially, the amount of sludge applied and the type of sludge (primary vs. secondary; percent solids) could affect the

TABLE 8-1

Factors and Weights Used to Evaluate
Potential for Microbiological Contamination of Groundwater[a]

Factor	Weights
Depth to Water (DTW)	5
Net Recharge (R)	2
Hydraulic Conductivity (K)	3
Temperature (T)	2
Soil Type (S)	5
Aquifer Medium (A)	3
Application Rate (AR)	4[b]
Distance to Well (D)	5

[a]Source: Developed by Yates, 1985

[b]Not used in the evaluation of landfills since expression was developed for application of sewage effluent.

quantity of virus or bacteria leached. However, insufficient information was available to develop a quantitative expression for these factors or their relative significance to the other factors. All other factors would essentially remain the same.

9. INFECTIOUS DOSE AND RISK OF DISEASE FROM MICROORGANISMS

9.1. INFECTIOUS DOSE

Important in any risk assessment is the level of concentration of contaminate that is necessary to cause an adverse effect on health.

Ideally, maximum contaminant levels for potentially harmful substances should be established on firm epidemiological evidence where cause and effect can be clearly quantified to determine a minimum- or no-risk level. However, while epidemiology is a valuable tool for detecting patterns of microbiological risk and establishing statistically significant associations with risk agents, it cannot quantitatively demonstrate cause and effect for pathogens (CST, 1983). Exact data on MID for humans are generally not possible because of the extreme cost, unethical nature of human experimentation and uncertainty in extrapolating dose-response curves to low exposure levels.

Risk assessment can be divided into four major steps: hazard identification, dose-response assessment, exposure assessment and risk characterization (NRC, 1983). The continuing occurrence of outbreaks of viral hepatitis and gastroenteritis in the United States clearly demonstrates that a hazard exists from viral contamination of drinking water.

The estimation of infective dose is difficult. To obtain data, which could be used for the purpose of predicting the probability of infection with low numbers of viruses, large numbers of individuals would be required who would have to be exposed to a highly virulent microorganism. Even if such experiments could be done, there would still be a great deal of uncertainty when extrapolating dose-response curves to low exposure levels. In addition, there are a number of factors that contribute to uncertainty in

determining MID. A number of these factors are listed in Table 9-1 along with an estimate of their contribution to uncertainty.

Ward and Akin (1984) recently reviewed the literature on MID of human viruses in a limited number of healthy individuals. The results indicated that relatively low numbers of viruses, perhaps 1 or 2 tissue culture PFU, were capable of causing infection.

A number of studies have been published in which small numbers of viruses, primarily vaccine strains, produced infection in human subjects. Koprowski et al. (1956) fed poliovirus 1 in gelatin capsules to adult volunteers and infected 2/3 subjects with 2 PFU of the virus. Katz and Plotkin (1967) administered attenuated poliovirus 3 (Fox) by nasogastric tube to infants and infected 2/3 with 10 $TCID_{50}$ and 3/10 with 1 $TCID_{50}$ of the virus. Minor et al. (1981) administered attenuated poliovirus 1 vaccine orally and infected 3/6 infants who were 2 months old with 50 $TCID_{50}$ of the virus.

The most extensive studies to date on MID for enteric viruses have been conducted by Schiff et al. (1984). Over 100 healthy adult volunteers were fed various doses of echovirus 12, a very mild pathogen, in drinking water. Using probit analysis, an estimated average MID of 17 PFU was obtained.

The infective dose of protozoan cysts also appears to be fairly low. The infective dose of Giardia lamblia and Entamoeba histolytica by the oral route appears to be between 1 and 10 cysts (Akin, 1983). Essentially one helminth egg can be considered to be infectious, although symptoms may be dose related (Kowal, 1985).

The MIDs for bacteria are generally higher than those for viruses and parasites. The number of ingested bacteria necessary to cause illness

TABLE 9-1

Contributors to Uncertainty in Determining

Minimum Infectious Dose for Enteric Viruses*

Category	Contribution to Uncertainty
1. Determination of immune status	One order of magnitude
2. Assay technique	One order of magnitude
3. Sensitivity of host	Several orders of magnitude
4. Virulence of virus	Several orders of magnitude
5. Use of upper 95% confidence limit	Up to one order of magnitude
6. Route of exposure	One order of magnitude
7. Choice of dose-response model	Several orders of magnitude
8. Synergism/antagonism	Many orders of magnitude
9. Dietary considerations	Uncertain
10. Distribution of subjects among doses and number used	1-2 orders of magnitude

*Source: Gerba, 1984a

appears to range from 10^2-10^8 (Akin, 1983). However, more recent studies suggest that the infective dose for Salmonella bacteria may be <10 organisms (D'Aoust, 1985). Virulence of the particular type and strain of organism as well as host factors may play a role in the actual number of organisms required to cause infection.

Unlike risks associated with toxic chemicals in water, individuals who do not actually consume or come into contact with contaminated water or sludge are also at risk. This is because microorganisms may also be spread by person-to-person contact or subsequent contamination of other materials with which noninfected individuals may come into contact. This secondary and tertiary spread of microorganisms has been well documented during water-borne outbreaks of infection caused by the Norwalk virus (Gerba et al., 1985). In the case of Norwalk outbreaks the secondary attack rate is ~30% (Figure 9-1).

9.2. ESTIMATED MORBIDITY AND MORTALITY FOR ENTERIC PATHOGENS

Not everyone who may become infected with enteric viruses or parasites will become clinically ill. Asymptomatic infections are particularly common among some of the enteroviruses. The development of clinical illness depends on numerous factors including the immune status of the host, age of the host, virulence of the microorganisms and type, strain of microorganism and route of infection. For hepatitis A virus (HAV) the percentage of individuals with clinically observed illness is low for children (usually <5%) but increases greatly with age (Evans, 1982) (Figure 9-2). In contrast, the frequency of clinical symptoms for rotavirus is greatest in childhood (Gerba et al., 1985) and lowest in adulthood. The observed frequencies of symptomatic infections for various enteroviruses are shown in Figure 9-3 (Cherry, 1981). The frequency of clinical HAV in adults is

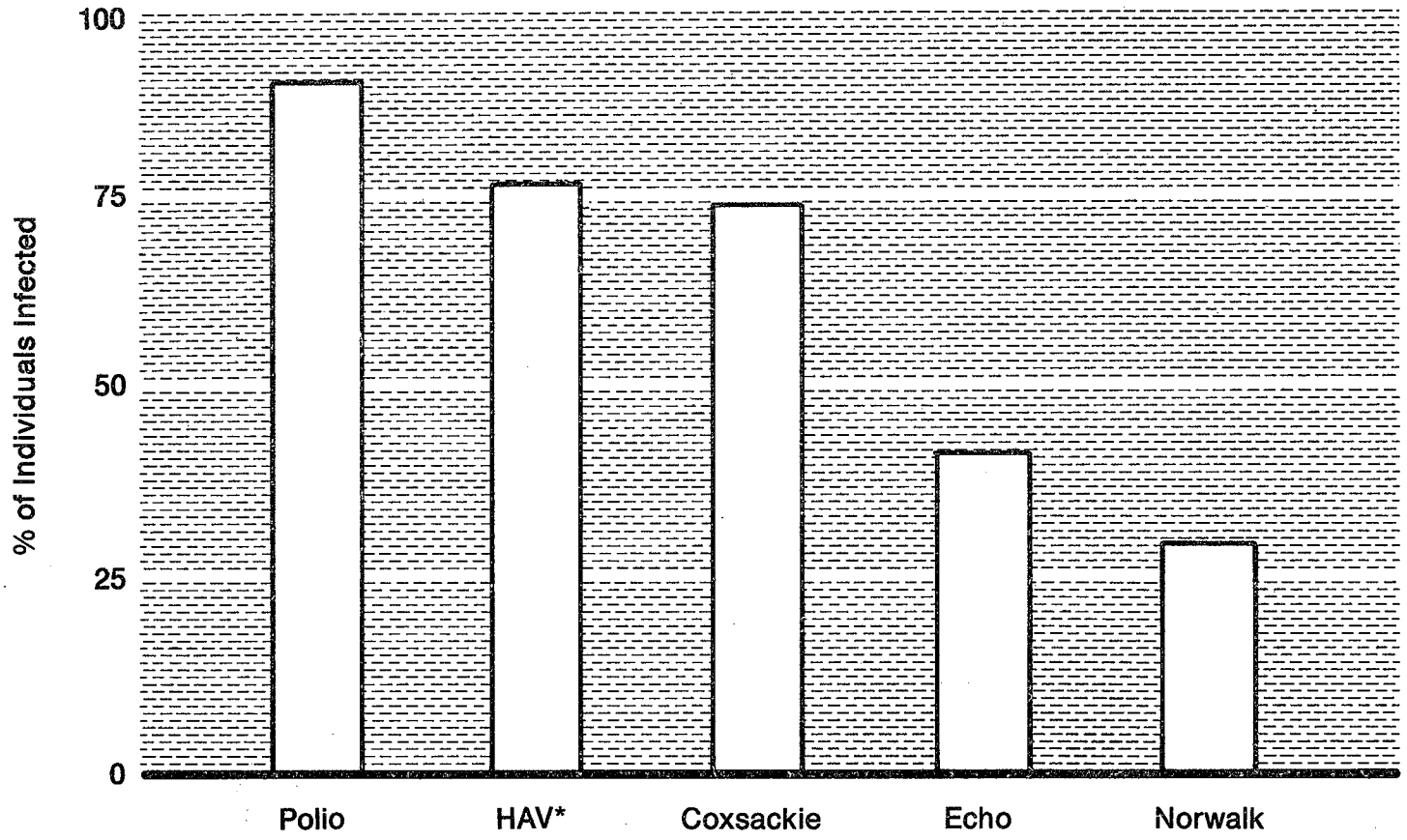


FIGURE 9-1

Secondary Attack Rates of Enteric Viruses

Source: Cherry, 1981; Evans, 1982; Gerba et al., 1985

*HAV -- Hepatitis A virus

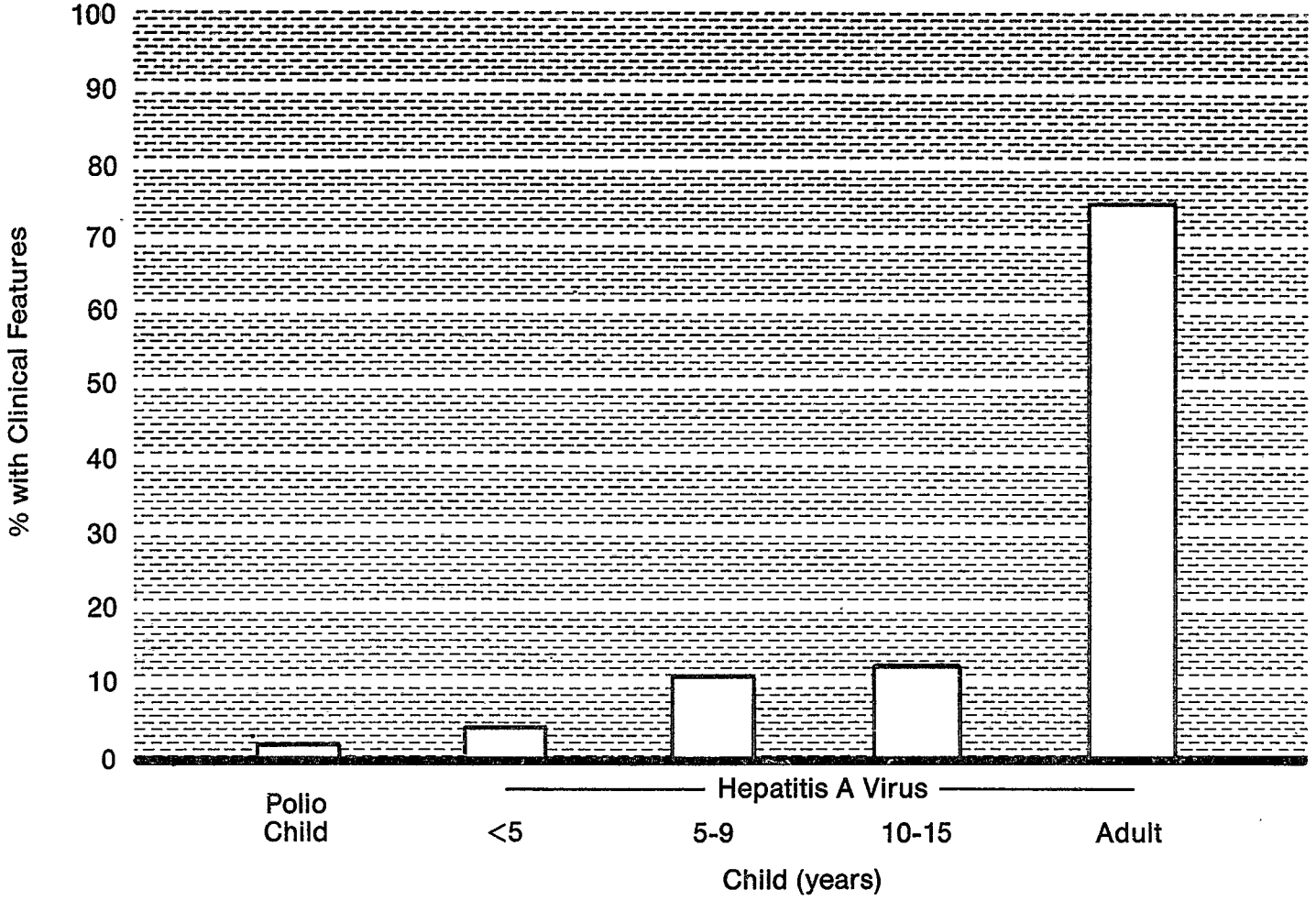


FIGURE 9-2

Percent of Individuals with Clinical Features for Polio and Hepatitis Virus by Age

Source: Evans, 1982

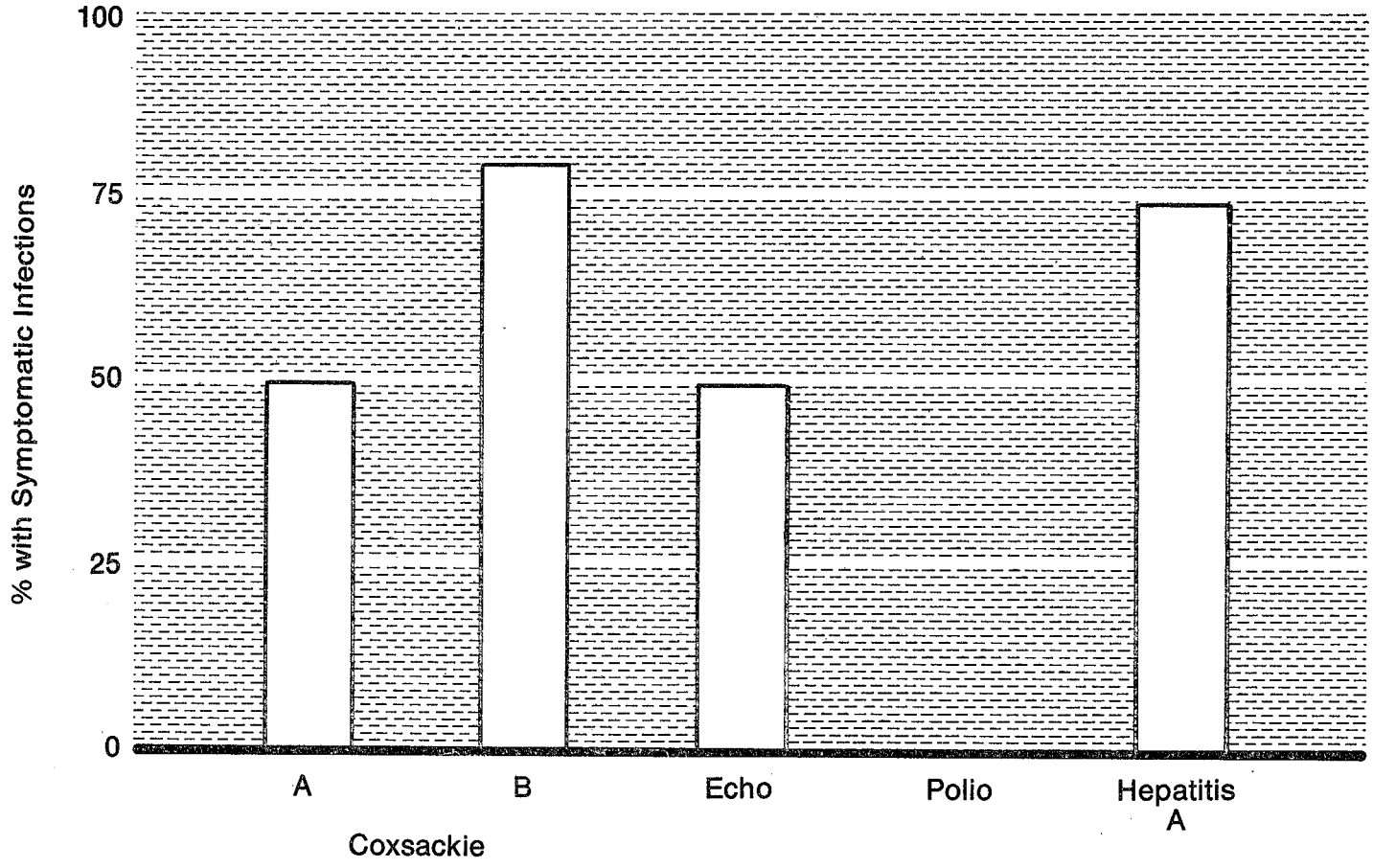


FIGURE 9-3

Frequency of Symptomatic Infections

Source: Cherry, 1981; Evans, 1982

estimated at 75%. However, during waterborne outbreaks of HAV it has been observed to be as high as 97% (Lednar et al., 1985).

Mortality rates are also affected by many of the same factors that determine the likelihood of the development of clinical illness. The mortality rate for salmonellosis in the United States is 0.2%, and shigellosis is 0.13% (Berger, 1986). The risk of mortality from HAV is 0.6% (CDC, 1985). Mortality from other enterovirus infections has been reported to range from <0.1%-1.8% (Assaad and Borecka, 1977). Mortality rates for enteric bacteria and enteroviruses are summarized in Table 9-2. The values for enteroviruses probably represent only hospitalized cases.

9.3. RISK ASSESSMENT FOR DRINKING WATER

The choice of extrapolation model is critical in any estimation of risk. Haas (1983a) compared the simple exponential model, a modified exponential model (beta) and the log-normal (or log-probit) model with experimental dose-response data. He concluded that the following model for viral infection (not necessarily disease) developed from the assumptions of random (Poisson) viral distribution and postingestion probability of viral infection that has a ß-distribution (Furomoto and Mickey, 1967) was superior at describing the data set:

$$P = 1-(1 + N/\beta)^{-a}$$

where

P = Probability of infection

N = Number of organisms ingested

and a and ß are parameters of distribution (Haas, 1983a).

Once the probability or risk of infection is determined, the annual and lifetime risks can be determined assuming a Poisson distribution of microorganisms within the water consumed (Haas, 1983b):

TABLE 9-2

Mortality Rates for Enteric Bacteria and Enteroviruses*

Organism	Mortality Rate (%)
Salmonella	0.2
Shigella	0.13
Hepatitis A	0.6
Coxsackie A2	0.5
A4	0.5
A9	0.26
A16	0.12
Coxsackie B	0.59-0.94
Echo 6	0.29
9	0.27
Polio 1	0.9

*Source: Assaad and Borecka, 1977; CDC, 1985; Berger, 1986. Data for polio, coxsackie and echo probably represent only hospitalized cases.

$$\text{Annual Risk} = 1(1-P)^{365}$$

$$\text{Lifetime Risk} = 1(1-P)^{25550}$$

The estimated annual risk of infection from one enterovirus in 1000 ℓ of drinking water (assuming ingestion of 2 ℓ/day), using infectivity data from different studies, is shown in Figure 9-4. Even if the highest infectious dose observed from human studies is used (Lepow et al., 1962), a significant risk of infection may result from low numbers of viruses in drinking water. Since the infectious dose of protozoan cysts appears to be similar to that of viruses (Akin, 1983), the risks could be presumed to be similar. Annual risks from bacteria appear to be substantially less; however, for Shigella and perhaps Salmonella bacteria (D'Aoust, 1985) there may be significant risk even when present in low numbers (Table 9-3).

Risks of mortality for some enteric pathogens also appear to be significant. Risks of mortality for Shigella dysenteriae, poliovirus 1, poliovirus 3 and HAV when present at different concentrations in tap water are shown in Tables 9-3 to 9-6. Risk of morbidity from one Shigella dysenteriae in 10 ℓ of drinking water could have a daily risk as high as 1.3×10^{-6}.

Risks for the enteroviruses, even considering that all infections do not result in clinical illness, also appear to be significant (see Tables 9-4 and 9-5).

The actual risk will always be underestimated since secondary and tertiary spread, which has been documented during waterborne disease outbreaks of Norwalk agent (Gerba et al., 1985), has not been included. Existing immunity tends to lower the risk, but recent studies with Norwalk agent (Blacklow et al., 1979) and echovirus 12 (Schiff et al., 1984) indicate that existing antibodies for these agents are not protective and multiple

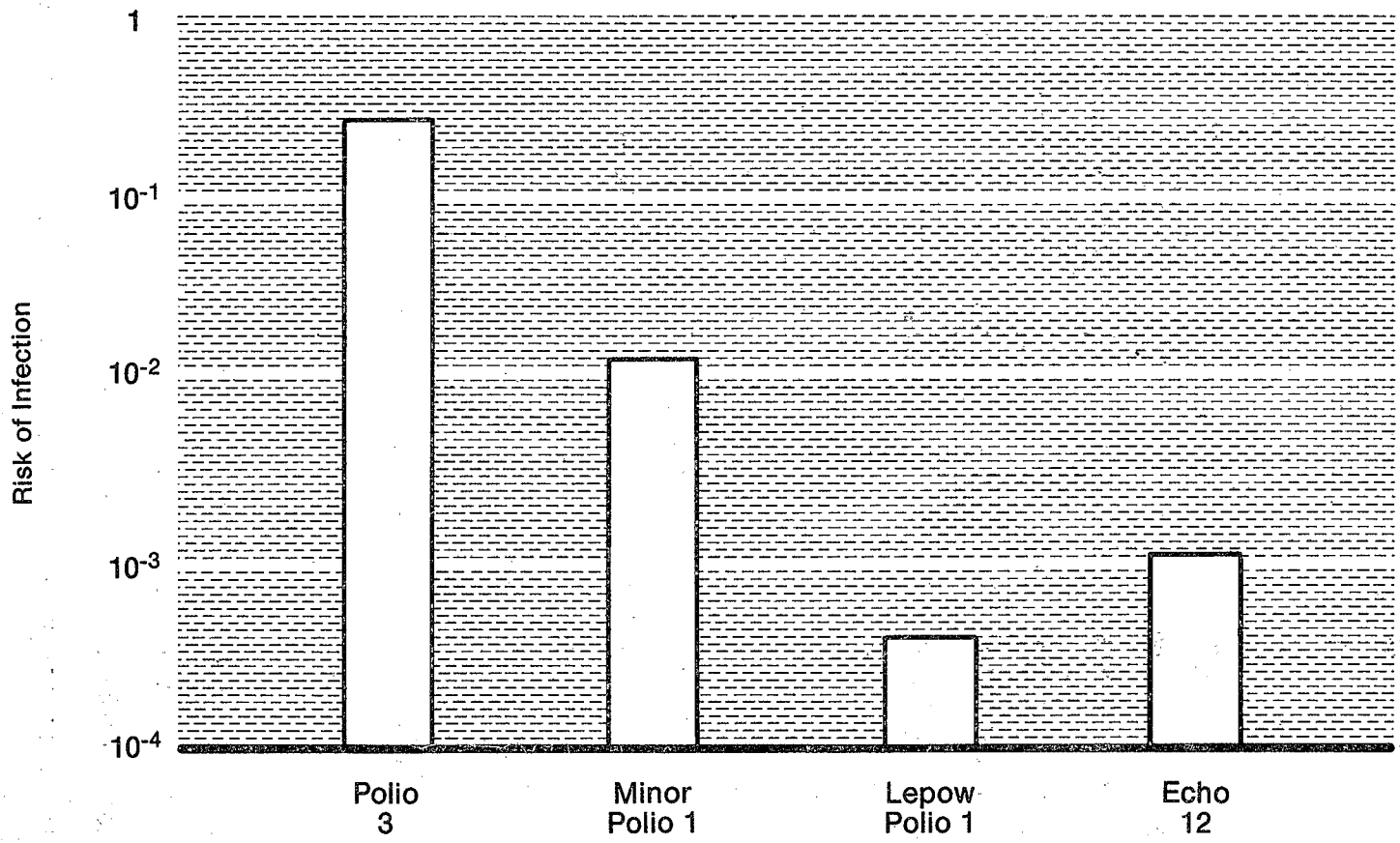


FIGURE 9-4

Annual Risk of Infection from 1 PFU/1000 ℓ

Source: Haas, 1983a

TABLE 9-3

Risk of Disease and Mortality from Concentrations of *Shigella dysenteriae* in Drinking Water*

Assuming One Virus in	Risk of Disease			Risk of Mortality		
	Daily	Annual	Lifetime	Daily	Annual	Lifetime
10 ℓ	1.0×10^{-4}	3.0×10^{-1}	1	1.3×10^{-6}	4.8×10^{-4}	3.3×10^{-2}
1000 ℓ	1.0×10^{-5}	3.6×10^{-3}	2.2×10^{-1}	1.3×10^{-8}	4.7×10^{-6}	3.3×10^{-4}
10,000 ℓ	9.5×10^{-7}	3.5×10^{-4}	2.4×10^{-2}	1.2×10^{-9}	4.4×10^{-7}	3.0×10^{-5}

*Source: Determined from data by DuPont and Hornick, 1973; Haas, 1983a; Berger, 1986

TABLE 9-4

Risk of Infection, Disease and Mortality from Various Concentrations of Poliovirus 1 in Drinking Water*

Assuming One Virus in	Daily	Annual	Lifetime
	Infection		
10 ℓ	3.0×10^{-3}	6.6×10^{-1}	1
1000 ℓ	2.9×10^{-5}	1.0×10^{-2}	5.2×10^{-1}
10,000 ℓ	3.6×10^{-6}	1.3×10^{-3}	8.7×10^{-2}
	Disease		
10 ℓ	3.0×10^{-6}	1.1×10^{-2}	5.3×10^{-1}
1000 ℓ	2.9×10^{-7}	1.1×10^{-4}	7.6×10^{-3}
10,000 ℓ	3.6×10^{-8}	1.3×10^{-5}	9.1×10^{-4}
	Mortality		
10 ℓ	2.7×10^{-7}	1.1×10^{-4}	7.6×10^{-3}
1000 ℓ	2.6×10^{-9}	9.5×10^{-7}	6.6×10^{-6}
10,000 ℓ	3.2×10^{-10}	1.1×10^{-7}	7.7×10^{-6}

*Source: Haas, 1983a; Minor et al., 1981

TABLE 9-5

Risk of Infection, Disease and Mortality from Various Concentrations of Poliovirus 3 in Drinking Water*

Assuming One Virus in	Daily	Annual	Lifetime
	Infection		
10 ℓ	8×10^{-2}	1	1
1000 ℓ	8.8×10^{-4}	2.7×10^{-1}	1
10,000 ℓ	4.4×10^{-5}	3.1×10^{-2}	8.9×10^{-1}
	Disease		
10 ℓ	8.0×10^{-4}	2.6×10^{-2}	1
1000 ℓ	8.7×10^{-6}	3.2×10^{-3}	2.0×10^{-1}
10,000 ℓ	8.8×10^{-7}	3.3×10^{-4}	3×10^{-2}
	Mortality		
10 ℓ	9.0×10^{-6}	3.3×10^{-3}	2.1×10^{-1}
1000 ℓ	9.6×10^{-8}	4.4×10^{-5}	3.0×10^{-3}
10,000 ℓ	9.6×10^{-9}	3.5×10^{-6}	2.5×10^{-4}

*Source: Haas, 1983a; Katz and Plotkin, 1967

TABLE 9-6

Risk of Infection, Disease and Mortality from Various Concentrations of Hepatitis A Virus in Drinking Water*

Assuming One Virus in	Daily	Annual	Lifetime
	Infection		
10 ℓ	2.9×10^{-3}	6.6×10^{-1}	1
1000 ℓ	2.9×10^{-5}	1.0×10^{-2}	5.1×10^{-1}
10,000 ℓ	3.5×10^{-6}	1.3×10^{-3}	8.7×10^{-2}
	Disease		
10 ℓ	2.2×10^{-3}	5.6×10^{-1}	1
1000 ℓ	2.1×10^{-5}	7.8×10^{-3}	4.2×10^{-1}
10,000 ℓ	2.7×10^{-6}	9.8×10^{-4}	6.6×10^{-2}
	Mortality		
10 ℓ	1.3×10^{-5}	4.9×10^{-3}	2.9×10^{-1}
1000 ℓ	1.2×10^{-7}	4.3×10^{-5}	3.0×10^{-3}
10,000 ℓ	1.6×10^{-8}	5.8×10^{-6}	4.1×10^{-4}

*Source: CDC, 1985; Minor et al., 1981; Haas, 1983a

infection and disease can occur. In contrast, existing antibodies for poliovirus and hepatitis clearly offer lifelong protection (Evans, 1982).

The actual distribution of pathogens in water is not known. The assumptions made in this risk assessment assume a random distribution (Haas, 1983b), which may or may not be the case. Additional research is necessary to determine the type of distribution in water for pathogens. This approach may also be utilized for pathogens in sludge.

9.4. SUMMARY OF DISEASE RISK FROM ENTERIC PATHOGENS

Many of the pathogens present in sludge are continuing causes of food and waterborne disease in the United States (Craun, 1986). While the information on infectious dose for most pathogens is limited, it appears that low numbers (<50 organisms) of viruses and protozoan cysts are capable of causing infection (Akin, 1983; Haas, 1983a) in a susceptible host. The number of individuals who develop clinical illness will depend upon the strain and type of organism as well as host factors such as age. The percent of individuals who develop clinical disease may be as low as 1% for poliovirus to as high as 97% for hepatitis. Significant mortality is associated with many of the viral pathogens, such as coxsackie and HAV, in some age groups.

If the distribution of pathogens in the environment (that is, water medium) is known, the risks of infection, morbidity and mortality can be estimated from existing data.

10. GROUNDWATER PATHWAY RISK ASSESSMENT METHODOLOGY

10.1. GENERAL ASSESSMENT

A major difficulty in assessing the risks of groundwater contamination by sludge-only landfills is the absence of any field or laboratory studies concerning the survival and transport of pathogens into groundwater by this method of municipal sludge disposal. Previous studies on land application of sludge have only been concerned with its application to the soil surface or within a few centimeters of the soil surface. Application rates at such sites are in the order of 22.4 metric tons/hectare vs. $\geq$22,417 metric tons/hectare at sludge landfills (U.S. EPA, 1978; Sagik et al., 1980). Thus, the concentration of pathogens/hectare is much greater at landfill sites.

A review of the literature suggests that, in terms of risk, significant concentrations of pathogens can be expected in the sludges that landfills receive. Most of the methods used in pathogen detection are not 100% efficient. In addition, methods do not exist for the detection of all of the pathogens that may occur in sewage sludges. As an example, recent studies on the occurrence of rotaviruses in anaerobically digested sludges suggest they occur in concentrations at least equal to that observed for the enteroviruses (Badawy, 1985) (see Table 5-4). It would be reasonable to suggest that the actual concentrations of enteric viruses are 10-100 times that observed experimentally.

It also appears that many of the pathogens are capable of prolonged survival in sludges, especially at low temperatures and high moisture. Indicator bacteria (coliforms and fecal coliforms) have been observed to survive for years in sludge and co-disposal landfills (Donnelly and

Scarpino, 1984). The high level of organic matter contributes to the survival and growth of indicator bacteria. Bacterial pathogens such as Salmonella are also capable of growth in sterilized sludges (Ward et al., 1984), although this appears unlikely in digested sludges because of the large number of antagonistic bacteria. Under ideal conditions viruses and parasites may survive for months to years, especially if subsurface temperatures approach 10°C.

Transport of pathogens from the sludge to the groundwater is more difficult to assess. The nature of the underlying soil is probably the most significant factor in controlling pathogen movement. In clayey soils or clay-lined landfills there is probably no movement of pathogens from the site. However, in larger grained soils at least some movement of pathogens could probably be assumed. No data base appears to exist to estimate the numbers of pathogens that could be leached from sludge landfills. The concentration of fecal coliform bacteria in sludge-only landfills has been reported to range from 2.4×10^3-2.4×10^4/100 mℓ and that of fecal streptococci from 2.1×10^3-2.4×10^5/100 mℓ (U.S. EPA, 1978). This suggests that significant leaching of pathogenic bacteria and viruses can occur. The chemical constituents of the leachate and its pH would be expected to have an influence on pathogen survival and transport (Chapters 6 and 7). The high organic content of the leachate (total organic carbon reported to be 10^2-1.5×10^4 mg/ℓ) could reduce viral and bacterial retention by soil particles as well as enhancing survival (U.S. EPA, 1978). Bacteria and viruses probably have the greatest chance of being leached from landfills. The amount of rainfall would probably be a major factor in microbial release from the sludge. In addition, the water content and weight of the sludge in landfills can be expected to increase water infiltration. Infiltration is also increased since the sludge provides

greater pore space and decreases the potential of surface sealing (Epstein, 1973). Sludges with a pH >7.0 would be expected to bind viruses less, so greater mobilization of viruses may occur. Studies with surface-applied sludges suggest that at least 0.1-1% of the viruses applied are released from the sludge (Ait et al., 1984). Numbers released may actually be greater since viral inactivation would be expected to be greater in surface-applied sludges because of drying and higher temperatures.

Any organisms released from the sludge would usually have to travel through an unsaturated zone before reaching the groundwater table. Removal of microorganisms in this zone is greater than the saturated zone (Chapter 7). Rainfall may play a significant role in the penetration of this barrier by microorganisms. Most of the landfills described in the U.S. EPA's Process Design Manual for Municipal Sludge Landfills (U.S. EPA, 1978) are constructed such that they are within 3 m of groundwater. While laboratory studies suggest substantial removal of microorganisms through the unsaturated zone, field studies indicate that penetration of enteric bacteria and viruses is possible. The degree of microbial removal will depend greatly upon the soil type. However, quantitative information on pathogen removal through the unsaturated zone is almost nonexistent (Chapter 7).

Less removal of microorganisms can be expected once they have entered the groundwater. Under saturated flow, viruses can travel long distances in sandy soils. The degree of removal is determined by the composition of the substrata. High removals can be expected in clay soils, while little removal probably occurs in fractured substrata or karst terrain. The rate of virus removal through soil observed in the laboratory differs from that observed in the field (see Tables 7-1 and 7-8). Laboratory studies with

poliovirus and echovirus suggest that 1-3 logs of virus removal would occur per meter of travel through sandy saturated loam soils. However, in field studies observed removals are usually <0.1 log/m. In one recent study, Stramer (1984) observed <0.05 log/m removal of poliovirus through silty loam soil under saturated flow conditions. The virus traveled over 46 m to contaminate a nearby lake. These results suggest that laboratory-grown viruses or laboratory experimental designs do not actually reflect virus transport through the subsurface in the field.

Bacteria and protozoa also appear capable of being transported several meters through sandy soils (Chapter 7). Giardia organisms can penetrate at least a meter of fine sand. Helminth eggs, because of their larger size, are unlikely to travel more than a few centimeters unless fractures in the substrata exist. Bacteria and protozoan cysts would not be expected to travel as great a distance in the subsurface as viruses.

10.2. CHARACTERISTICS OF BEST- AND WORST-CASE LANDFILLS

Based on a review of the literature, the best and worst landfill sites as far as potential risks for contamination of groundwater are shown in Table 10-1. The ideal site would utilize digested secondary sludge with a solids content of $\geq$20%. The substrata would be a clayey soil with a deep groundwater table and in an area of low rainfall. With a clayish soil and a clay lining, no enteric pathogens would be expected to contaminate groundwater. A worst-case landfill would dispose of raw or primary sludge with a solids content of <15%, lie within 1 m of the groundwater table, be unlined with a sand to gravel substrata and in an area of high rainfall.

Also presented in Table 10-1 are representative conditions present at 15 sludge landfill sites reviewed in the U.S. EPA's Process Design Manual on Municipal Sludge Landfills (U.S. EPA, 1978). A review of the characteris-

TABLE 10-1

Characteristics of Best, Worst and Average Operated Sludge Landfills

Example	Sludge Characteristics: Treatment	Sludge Characteristics: Percent Solids	Sludge to Groundwater (meters)	Site Characteristics: Groundwater Flow (meters/day)	Site Characteristics: Nature of Soil	Site Characteristics: Rainfall (centimeters)	Site Characteristics: Groundwater Temperature (°C)	Other
Best	stabilized, anaerobic digestion, lime dewatered	20	≥30.5	0.3	clay	50.8	30	clay lined area fill
Operated landfills*	stabilized, raw, anaerobic digested, dewatered	3-28	6.7	--	clay to gravel	25.4-101.6	3-17	clay lined to unlined
Worst	unstabilized, raw	15	≤0.9	≥3.0	gravel-sand	101.6	10	unlined

*Source: U.S. EPA, 1978

1 meter = 3.281 feet
1 centimeter = 0.3937 inches

tics of these sites suggests that both raw and stabilized sludges are disposed, that the depth to groundwater is often within 1 m, the substrata is clay to gravel and rainfall is >76.2 cm/yr. Some sites are clay lined.

10.3. EVALUATION OF GROUNDWATER CONTAMINATION AT LANDFILLS BY MICRO-DRASTIC

Two example sludge landfill sites were evaluated using the micro-DRASTIC rating system developed by Yates (1985) to assess the likelihood of groundwater contamination. The site characteristics used in the micro-DRASTIC system for each of the two sites is shown in Table 10-2. The application rate factor was not used since liquid wastes were not applied to these sites. The ratings developed by Yates (1985) were used to determine each factor (Tables 10-3 and 10-4). Scores were determined by multiplying the ratings by the weights in Table 10-5. The total scores and ratings for directly beneath the site and distances of 100 m and 200 m are shown in Table 10-6. Using the suggested rating scale of Yates (1985) (Table 10-7), it was determined that microbial contamination was probable directly beneath Site A and possible beneath Site B. At both sites it was judged to be possible at distances of 100 and 200 m.

It should be pointed out that the rating system is attributary and has not been verified in the field. Still, it provides a mechanism for evaluating the many interacting factors controlling microbial survival and transport in the subsurface. While the rating system suggests that microbial contamination is possible at all sites, it does not necessarily mean that microbial contamination will occur. It does suggest that, based on the current body of information on microbial behavior in the subsurface, it cannot be excluded at the present time. Micro-DRASTIC could potentially be used like DRASTIC as a first step in evaluating the potential for microbial

TABLE 10-2

Characteristics of Selected Municipal Sludge Landfills

Site	Depth to Water (m)	Net Recharge[a] (m)	Hydraulic Conductivity[a] (ℓpd-m²)	Temperature[b] (°C)	Soil Type	Aquifer Medium	Sludge Type
A	3-4	0-1	0.35	10	clay, sand and gravel	silt loam	secondary
B	10-12	7-8	0.35-0.035	14	silty clay	silty	primary

[a]Values obtained from Aller et al., 1985

[b]Values of annual mean temperature from DOC-NOAA (1982) for typical sludge landfill sites in the United States

1 liter = .2642 gallons
1 square meter = 10.76 square feet

TABLE 10-3

Rating of Microbial Contamination at Sludge Disposal Site A

Factor	Directly Beneath Site		100 m		200 m	
	Rating[a]	Score[b]	Rating	Score	Rating	Score
Depth to ground-water	10	50	10	50	10	50
Net recharge	1	2	1	2	1	2
Hydraulic conductivity	1	3	1	3	1	3
Temperature	9	18	8	18	9	18
Soil type	7	35	7	35	7	35
Aquifer medium	10	30	5	15	4	12
Distance	10	50	3	15	2	10
Total Score		188		138		130

[a]Source: Yates, 1985

[b]Score is obtained by multiplying the weights shown in Table 10-5 by the rating.

TABLE 10-4

Rating of Microbial Contamination at Sludge Disposal Site B

Factor	Directly Beneath Site		100 m		200 m	
	Rating[a]	Score[b]	Rating	Score	Rating	Score
Depth to groundwater	9	45	9	45	9	45
Net recharge	7	14	7	14	7	14
Hydraulic conductivity	1	3	1	3	1	3
Temperature	9	18	9	18	9	18
Soil type	2	10	2	10	2	10
Aquifer medium	4	12	2	6	2	6
Distance	10	50	3	15	2	10
Total Score		152		111		106

[a]Source: Yates, 1985

[b]Score is obtained by multiplying the weights shown in Table 10-5 by the rating.

TABLE 10-5

Factors and Weights Used to Evaluate
Potential for Microbiological Contamination of Groundwater*

Factor	Weights
Depth to Water (DTW)	5
Net Recharge (R)	2
Hydraulic Conductivity (K)	3
Temperature (T)	2
Soil Type (S)	5
Aquifer Medium (A)	3
Distance to Well (D)	5

*Source: Yates, 1985

TABLE 10-6

Estimation of Microbial Contamination at Two Sludge Disposal Sites*

Site	Total Score at Distance Indicated			Estimation of Microbial Contamination		
	Directly Beneath	100 m	200 m	Directly Beneath	100 m	200 m
A	188	138	130	probable	possible	possible
B	152	111	106	possible	possible	possible

*Values developed for site examples from rating system of Yates (1985).

TABLE 10-7

Volume of Sludge and Net Recharge per Year at Sludge Landfills A and B*

Site	Type of Sludge	Dry Metric Tons/ Cubic Meter	Cubic Meters/ Hectare	Metric Tons/ Hectare	ℓ/Hectare	Net Recharge (centimeters/ hectare)	Net Recharge (ℓ/hectare)
A	secondary	0.15	17,004	2694.5	1.1×10^9	6.3	253,667
B	primary	0.23	17,193	3875.8	1.2×10^{10}	50.2	2,029,335

* Values developed for site examples from rating system of Yates (1985).

1 metric ton = 1.1023 short tons
1 cubic ton = 1.308 cubic yards
1 hectare = 2.471 acres
1 liter = 0.2642 gallons
1 centimeter = 0.3937 inches

contamination for a particular site based upon hydrogeologic settings. To be more useful, ratings for sludge type (or any additional treatments, for example, composting) and application rate could be included.

10.4. ESTIMATING TRANSPORT OF ENTERIC ORGANISMS AT SLUDGE LANDFILLS TO GROUNDWATER

Another approach to determine the likelihood of groundwater contamination is to estimate the leaching of pathogens from landfills and their concentrations at various distances from the sites. From this information, risk of illness by using the water for drinking could be estimated by the methods shown in Chapter 9. To determine the number of pathogens reaching any location beneath or distant from the site, the number of pathogens is first estimated. This will be dependent on the type of sludge and the treatment(s) it has received. The number of pathogens leached from the landfill is then determined and their expected rate of removal as they travel through the unsaturated zone. Once the pathogens have reached the groundwater (saturated zone) their removal will be less than in the unsaturated zone. This method is diagrammed in Figure 10-1.

To illustrate how this methodology might be used, two example sites were chosen, one disposing of primary sludge and the other secondary sludge. The volume of sludge disposed at each site and net recharge are shown in Table 10-7. From the net recharge the volume of leachate, which could be generated per hectare, can be determined (that is, the volume of recharge rainwater that passes through the saturated zone per hectare). From the literature review (Chapter 5) the number of pathogens expected per hectare at each landfill site was estimated and is shown in Table 10-8.

Whether or not a pathogen reaches groundwater and is transported to drinking-water wells depends on a number of factors including initial con-

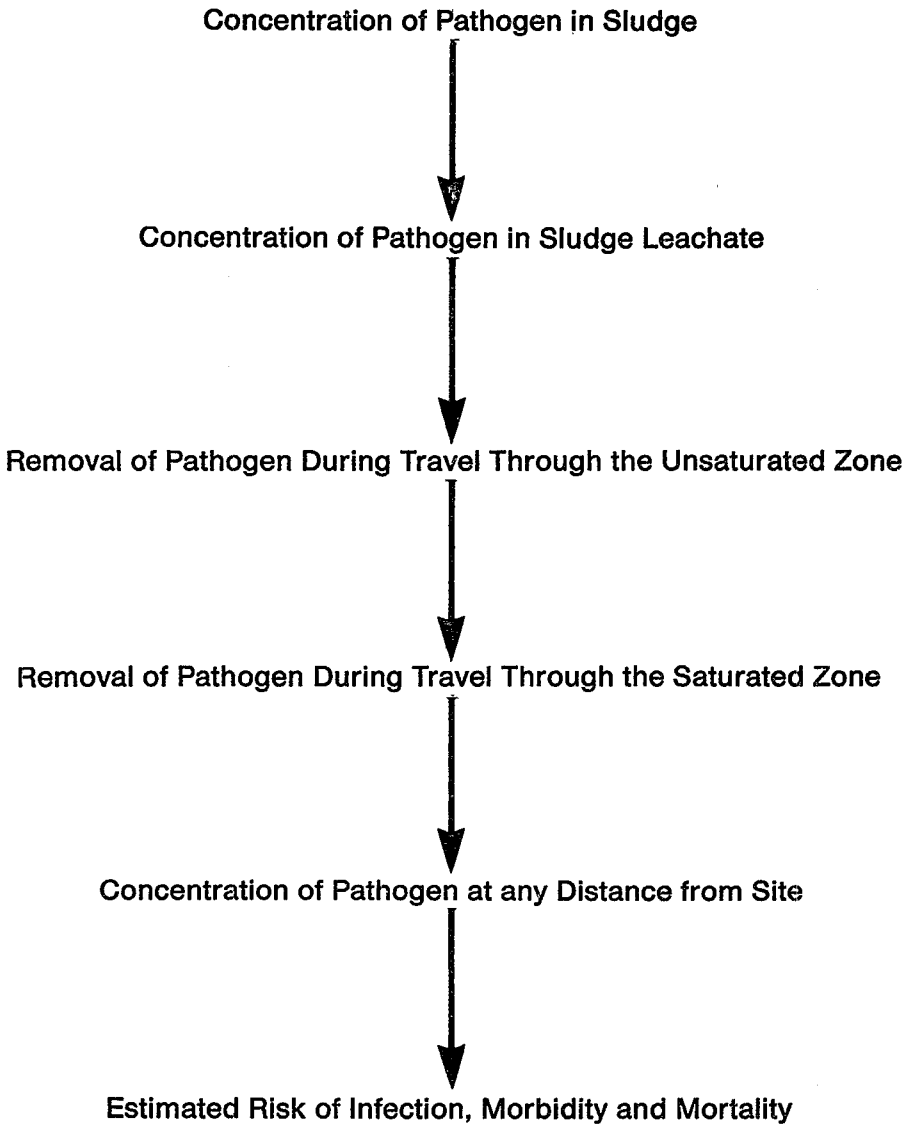


FIGURE 10-1

Methodology for Estimating Risks from Groundwater Contamination by Sludge Landfills

TABLE 10-8

Estimated Levels of Pathogens and Indicator Bacteria at Sludge Landfills A and B Applied/Hectare*

Pathogen	Site A	Site B
Enteroviruses	10^{3}-10^{4}	$10^{4.4}$-10^{5}
Rotavirus	$10^{3.6}$-10^{4}	unknown
Salmonella	10^{3}-$10^{4.4}$	$10^{4.4}$-$10^{4.8}$
Fecal Coliforms	10^{4}-$10^{5.6}$	$10^{6.4}$-$10^{6.8}$
Ascaris	$10^{3.6}$	$10^{4.4}$-$10^{4.8}$

* Based on literature review contained in this document.

centration of the pathogens, survival of the pathogens, number of pathogens that reach the sludge-soil interface, the degree of removal through the unsaturated and saturated soil zones and the hydraulic gradient. The degree to which each of these factors will influence the probability of pathogens entering groundwater cannot be determined precisely. Viruses, because of their small size, probably have the greatest potential of all the pathogens of actually reaching the groundwater and being transported from the site. To determine the numbers of viruses that may be transported from a site, assumptions were made for each of the principal factors (Table 10-9). Values were estimated for most favorable, most probable and worst possible conditions. Most favorable conditions are those conditions most likely to result in limited virus survival and transport (Table 10-10).

10.4.1. Estimated Concentration of Viruses in the Sludge. The observed ranges of enteroviruses and rotaviruses detected in sewage sludges are discussed in Chapter 5. The concentration of other enteric viruses such as Norwalk, hepatitis A and adenovirus is not known, but obviously, they will also be present. Limited studies on the presence of rotaviruses suggest that they will also be present in numbers at least equal to that observed for enteroviruses in anaerobically digested sludge. However, detection methods for even the enteroviruses in sludge are only 30-50% efficient (Farrah and Schaub, 1983). Thus, the actual number of pathogenic enteric viruses is undoubtedly many times that detected by conventional methods. Modification of tissue culture techniques can result in a 10-fold or greater increase in the numbers of enteroviruses detected in sewage (Morris and Waite, 1980). For most favorable conditions the number of enteric viruses was placed at 14/g of sludge, which is the combined total observed for

TABLE 10-9

Assumptions Used in Assessing Virus Contamination of Groundwater at Sludge Landfills

Item (unit of measurement)	Most Favorable	Most Probable	Worst Possible
Concentration of viruses in sludge:			
Secondary (g)	14	3500	7×10^4
Primary (g)	100	10^4	10^6
Percent of viruses leached from sludge	0.1	1	10
Concentration of viruses in leachate (ℓ)	Viruses are suspended in total volume of net recharge.		Viruses are only eluted slowly or only after significant rainfall through a limited number of pores. Viruses are resuspended in only 1% of the total volume of leachate.
Removal rate of viruses through unsaturated zone (m)	2 logs	0.1 log	0.006 log
Removal rate of viruses in saturated zone (m)	2 logs	0.1 log	0.006 log
Inactivation or decay constant (k)	0	0	0
Rate of travel (m/day)	Since no inactivation (decay) is assumed, this factor is not important in the risk assessment.		
Dispersion and dilution	0	0	0

TABLE 10-10

Basis of Assumptions Used in Assessing Contamination of Groundwater at Sludge Landfills by Viruses

Item	Most Favorable	Most Probable	Worst Possible
Concentration of viruses in sludge	Combined lowest reported concentrations of entero-viruses and rotaviruses	Combined median-range concentrations reported for enteroviruses and rotaviruses times 10	Combined highest reported concentrations for entero-viruses and rotaviruses times 100
Percent of viruses leached from sludge	That observed from surface spreading of sludge on land	10 times that of most favorable	100 times that of most favorable
Concentration of viruses in leachate	Viruses are suspended in total volume of net recharge	Same as most favorable	Viruses are suspended in 1% of the volume of the net recharge
Removal rate	That observed for sandy loam soils in laboratory studies with enteroviruses	Median of that observed for field studies with enteroviruses	Lowest of that for field studies with enteroviruses
Inactivation or decay constant	That observed at 10°C or below (no inactivation)	Same as most favorable	Same as most favorable
Rate of travel	Average groundwater flow	Based on laboratory study (Grondin and Gerba, 1986)	Same as most probable
Dispersion and dilution	This is a very site-dependent factor and is not considered in the risk assessment		No dilution or dispersion

enteroviruses and rotaviruses and is based on the lowest concentration of enteroviruses that has been observed. Most probable represents the combined median-range concentrations observed increased by a factor of 10 to take into consideration limits of detection methods. Worst possible is the highest concentration observed for enteroviruses times a factor of 100. This is considered to be the highest concentration likely to be present.

10.4.2. Percent of Viruses Released from Sludge. Quantitative studies have never been conducted on the degree to which viruses can be leached from sludges during water infiltration. Laboratory studies have been conducted for surface application of sludges, but inactivation from dessication would likely reduce the numbers of enteroviruses rapidly under these conditions (see Chapter 7). In addition, these experiments were done with acid soils, which would act to reduce virus migration through adsorption. However, field studies indicate that transport to groundwater occurs under field conditions where sandy soils exist. The laboratory studies suggest that at least 0.1-1% of the viruses present may be leached from the sludge. However, greater numbers could be leached from buried sludges since dessication will be less than would occur in surface-applied sludges. Under most favorable conditions, it is estimated that no more than 0.1% of the viruses are released from the sludge. For most probable, 1% is estimated, and under worst possible a 10% release is estimated.

10.4.3. Concentration of Viruses in Sludge Leachate. It is difficult to estimate the volume of water at which the viruses will be suspended after they are leached from the buried sludge. The field studies of Wellings et al. (1974) suggest that viruses may only be eluted from soils and penetrate to groundwater after significant rainfall. If true, then viruses may only

be released from the sludge in "bursts" after a major rainfall rather than slowly as rainfall migrates through the sludge. Under most favorable and most probable conditions it is estimated that viruses are released slowly into the net recharge volume at a sludge landfill. For worst possible conditions viruses are only released in 1% of the net recharge volume.

10.4.4. Removal Rate by Soil. Laboratory studies suggest that most viruses are rapidly removed from water by soil during percolation. Greater removal would be expected during movement through the unsaturated zone than the saturated zone (Chapter 7). Little information is available on virus movement through the unsaturated zone. Removal through sandy soils appears to be in the order of 1-3 logs/m. However, field studies suggest that under "real-life" conditions virus removal is 1-2 orders of magnitude less. Little is known on factors governing virus survival and transport under field conditions. The most favorable situation relies on laboratory data for sandy soils that suggest 2-log removal of virus/m of soil. For favorable and probable conditions removal is estimated at 0.1 log/m based on average field conditions (see Table 7-8). Worst possible is the lowest removal under field conditions observed for silty soil conditions. Since field studies have not yet shown any differences between virus removal in unsaturated vs. saturated zones, the rate of removal is assumed to be the same for both zones in the subsurface (Chapter 7).

10.4.5. Inactivation or Decay Rate of Viruses. Previous studies have shown that inactivation (decay) rates of viruses can be estimated from the median temperature of the groundwater (Yates, 1985). Recent research suggests that hepatitis A virus is substantially more resistant to thermal inactivation than all other enteric viruses in groundwater (Sobsey, 1985).

Research currently in progress will provide this information so that more accurate estimations can be made. Unfortunately, no data are available on viral survival in landfill sludge leachates. The leachate composition could have a major impact on viral inactivation rates. The high organic content could act to retard virus inactivation (see Table 6-4). Because of the lack of information on what virus inactivation would be in sludge leachate, it is assumed that it would be zero for all conditions. This would be the case anyway if groundwater temperatures were at or near 10°C (Yates, 1985). This is not an unrealistic assumption since many sludge landfills are located in areas where the groundwater temperature would be in this range (U.S. EPA, 1978; Yates and Gerba, 1985).

10.4.6. Rate of Travel, Dilution and Dispersion. In determining the concentration of viruses at a given point from a landfill, it is necessary to determine the time required so that the inactivation (decay) rate of the viruses can be taken into consideration. The application of proposed models for virus transport (Yates et al., 1986) would be useful in predicting rate of travel, dispersion and dilution. However, current models have not been verified by laboratory and field studies. A recent laboratory study suggests that viruses may travel at 1.5-1.9 times faster than the average flow of groundwater (Grondin and Gerba, 1986), which implies that current solute models may not be totally applicable to modeling the movement of microorganisms.

For this risk assessment information on rate of travel is not needed since no virus inactivation is assumed to occur. Since no information is available to verify proposed models, no dilution or dispersion is assumed to occur.

10.4.7. Risk Assessment at Example Sludge Landfill A. Using the previously discussed assumptions, the concentrations of viruses in the sludge leachate and at 10 m and 100 m from the site were estimated (Table 10-11).

10.4.8. Risk Assessment at Example Sludge Landfill B. Primary sludge is disposed at Site B so the estimated number of pathogens is much greater. Again, under most favorable conditions, only the sludge leachate would contain significant numbers of viruses (Table 10-12). Highly significant risk of infection (Chapter 9) would exist from groundwater use $\leq$10 m from the site under most probable and worst possible conditions. Risks would also be significant at 100 m from the site with both assumptions. The higher net recharge at Site B results in a greater dilution of the viruses than at Site A. Risks would be greater at Site B if rainfall were similar to Site A.

10.5. SUMMARY OF GROUNDWATER RISK ASSESSMENT AND RESEARCH NEEDS

Two approaches were used to determine if sludge landfills pose a risk to groundwater. In one approach two example landfill sites were evaluated using Yates' rating system, and in the other approach an attempt was made to estimate the numbers of pathogens that would leach from the sludge at two example sites and find their way into the groundwater. The results suggest that contamination of the groundwater is possible directly beneath sludge landfill sites as well as at a distance. In terms of potential disease and mortality from consumption of the water near these sites, the risks appear significant based on annual and lifetime water use (see Tables 9-4 through 9-6). The example sites are underlaid with silty loam soils but some sites in the United States are built upon sandy soils. If these sites are not clay lined, greater risks could be expected from microbial contamination than the example sites examined in this assessment. Little risk exists from clay-lined sludge landfills.

TABLE 10-11

Estimated Concentrations of Viruses in Groundwater at Sludge Landfill A[a]

Item	Most Favorable	Most Probable	Worst Possible
Virus/hectare	2.7×10^{4}	4.2×10^{4}	1×10^{5}
Viruses in leachate/hectare	1.7×10^{2}	4.2×10^{4}	1×10^{5}
Virus concentration in leachate/ℓ	1.7×10^{1}	4.1×10^{5}	9.8×10^{8}
Virus concentration[b] 10 m from site/ℓ	1.7×10^{-25}	2.0×10^{4}	8.1×10^{8}
Virus concentration 100 m from site/ℓ	---	2.0×10^{-5}	2.0×10^{8}

[a]Secondary sludge

[b]After travel through 3 m of unsaturated soil (total distance 13 m)

TABLE 10-12

Estimated Concentrations of Viruses in Groundwater at Sludge Landfill B[a]

Item	Most Favorable	Most Probable	Worst Possible
Virus/hectare	1.0×10^{5}	1.0×10^{6}	1.0×10^{6}
Viruses in leachate/hectare	1.0×10^{4}	1.0×10^{5}	1.0×10^{6}
Virus concentration in leachate/ℓ	1.2×10^{3}	1.2×10^{6}	1.2×10^{10}
Virus concentration[b] 10 m from site/ℓ	1.2×10^{-23}	6.0×10^{4}	1.0×10^{16}
Virus concentration at 100/m from site/ℓ	---	6.0×10^{5}	1.2×10^{9}

[a]Primary sludge

[b]After transport through 3 m of unsaturated soil (total distance 13 m)

If actual drinking-water wells had existed near these sites, the potential risks would have been greater than that determined. Since pumping wells can greatly influence the movement of water, they would be expected to enhance water movement under a site and reduce microbial removal efficiency.

It is clear from this review that information on the fate of pathogens at existing landfills is essentially nonexistent. Both laboratory and field studies are needed to determine the degree of pathogen leaching, survival and transport in groundwater. Approaches are available to estimate potential risks from pathogens at sludge landfills, but without adequate information the reliability of the conclusions is weakened. The availability of necessary information to make a risk assessment and research needs are shown in Table 10-13.

TABLE 10-13

Status of Information for Groundwater Risk Assessment and Research Needs

Item	Adequate Data to Make a Risk Assessment				Research Needs
	Bacteria	Viruses	Protozoan Cysts	Helminth Eggs	
Concentration in sludge	yes	yes	yes	yes	Data for emerging pathogens and better detection methods needed
Concentration of organisms leachated	limited	no	no	no	No data on pathogens. Field and laboratory studies needed
Survival (decay rate) in leachate	no	no	no	no	No data on pathogens
Survival (decay rate) in groundwater	yes	yes	limited	limited	Data for emerging pathogens would be useful
Transport through unsaturated zone	limited	limited	limited	limited	Limited data available. Information on effect of rainfall needed
Transport through saturated zone	yes	limited	limited	yes	Proposed models need laboratory and field verification
Risk of illness	yes	yes	yes	yes	Data needed on nature of distribution of pathogens in water

11. SUMMARY AND CONCLUSIONS

The purpose of this document was to identify human pathogens associated with sewage sludge and the risks posed by such pathogens following the disposal of sludge in municipal landfills. Background information on pathogens of concern and their persistence in various landfill environments has been presented. Attempts have also been made to identify different routes by which pathogens disposed of in municipal sludge landfills can reach humans and to estimate risks associated with each of the potential routes.

Survival characteristics of pathogens are critical factors in assessing risks associated with potential transport of microorganisms from soil-sludge to the groundwater environments of landfills, and this document presented and discussed various modes for predicting microbial die-off. Temperature is probably even more important than pH and moisture content in predicting pathogen survival. The order of persistence in the environment from longest to shortest survival time appears to be helminth eggs > viruses > bacteria > protozoan cysts.

Factors affecting pathogen movement in the sludge-soil matrix include physical characteristics of the soil as well as environmental and chemical factors. In most soils, viruses could be expected to travel farthest because of their small size, while the movement of protozoa and helminths would be more limited. However, many other factors, including rainfall and soil inhomogeneities, have major impacts on pathogen movement under field conditions. The depth of the unsaturated zone is probably the greatest barrier preventing microbial movement into the groundwater.

MIDs for bacteria are generally higher than those for viruses and parasites. Mathematical modeling indicates that <50 viruses or protozoan cysts are capable of causing infection in a susceptible host.

A methodology known as micro-DRASTIC, developed to evaluate the potential for groundwater contamination of septic tanks by microorganisms based on eight key rating factors, was applied to assess the likelihood of

groundwater contamination at two example sludge landfill sites. It was determined that microbial contamination was probable directly beneath the first site (secondary sludge; clay, sand, and gravel; silt loam aquifer) and possible beneath the second site (primary sludge, silty clay, silty aquifer medium). At both sites, pathogenic contamination was judged to be possible at distances of 100 m and 200 m. Although this rating system has not been verified in the field, it provides a mechanism for evaluating the many interacting factors that control microbial survival and transport in the subsurface.

Whether or not a pathogen reaches groundwater and is transported to drinking-water wells depends upon a number of factors, including initial concentration of the pathogen, survival of the pathogen, number of pathogens that reach the sludge-soil interface, degree of removal through the unsaturated and saturated soil zones, and the hydraulic gradient. The degree to which each of these factors will influence the probability of pathogens entering groundwater cannot be determined precisely. Viruses, because of their small size, probably have the greatest potential of all the pathogens of actually reaching the groundwater and being transported from the site.

Information on the fate of pathogens at existing landfills is sorely lacking. Additional laboratory and field studies are needed to determine the degree of pathogen leaching, survival and transport in groundwater in order to estimate potential risks from pathogens at sludge landfills with reasonable validity.

12. REFERENCES

Ait, N., M. Afanassieff, D. Garcia, et al. 1984. Virus binding in sludge and soil. In: Processing and Use of Sewage Sludge, P. L'Hermite and H. Ott, Ed. Reidel, Dordrecht, Holland. p. 224-236.

Akin, E.W. 1983. A review of infective dose data for enteroviruses in human subjects. In: Microbial Health Considerations of Soil Disposal of Domestic Wastes. U.S. EPA, Cincinnati, OH. EPA-600/9-83-017. p. 304-317.

Akin, E.W., W.H. Benton and W.F. Hill. 1971. Enteric viruses in ground and surface water: A review of their occurrence and survival. In: Virus and Water Quality: Occurrence and Control, V. Snoeyink, Ed. Univ. of Illinois, Urbana. p. 59-73.

Allen, M.J. and S.M. Morrison. 1973. Bacterial movement through fractured bedrock. Ground Water. 11: 6-10.

Aller, L., T. Bennett, J.H. Lehr, et al. 1985. DRASTIC: A Standardized System for Evaluating Groundwater Pollution Potential Using Hydrogeologic Settings. U.S. EPA, Ada, OK. EPA/600/2-85/018.

Anderson, B.C. 1985. Moist heat inactivation of Cryptosporidium sp. Am. J. Public Health. 75: 1433-1434.

Angus, K.W. 1983. Cryptosporidiosis in man, domestic animals and birds: A review. J. R. Soc. Med. 76: 62-69.

APHA (American Public Health Association). 1975. Control of Communicable Diseases in Man, 12th ed. APHA, Washington, DC.

Aserkoff, B., S. Schroeder and P. Brachman. 1970. Salmonellosis in the United States -- A five-year review. Am. J. Epidemiol. 92: 13-24.

Assaad, F. and I. Borecka. 1977. Nine-year study of WHO virus reports on fatal virus infections. Bull. W.H.O. 55: 445-453.

Aulenbach, D.B., T.P. Glavin and J.A.R. Rojas. 1974. Protracted recharge of treated sewage into sand. Part I: Quality changes in vertical transport through the sand. Ground Water. 12: 161-169.

Badawy, A. 1985. Unpublished results. Univ. of Arizona, Tucson, AZ.

Beard, P. 1940. Longevity of E. typhosa in various soils. Am. J. Public Health. 30: 1077-1082.

Beaver, P.C. and G. Deschamps. 1949. The viability of Entamoeba histolytica in soil. Am. J. Trop. Med. Hyg. 29: 189-191.

Berger, P. 1986. U.S. EPA, Washington, DC. Memorandum to C. Gerba, Univ. of Arizona, Tucson, AZ. January 1. Data obtained from the Centers for Disease Control, Atlanta, GA.

Bingham, A.K., E.L. Jarroll and E.A. Meyer. 1979. Induction of Giardia excystation and the effect of temperature on cyst viability as compared by eosin-exclusion and in vitro excystation. In: Waterborne Transmission of Giardiasis, W. Jakubowski and J.C. Hoff, Ed. U.S. EPA, Cincinnati, OH. EPA-600/9-79-00. p. 217-229.

Bitton, G. and C. Gerba. 1984. Groundwater Pollution Microbiology. John Wiley and Sons, New York.

Bitton, G., O.C. Pancorbo, A.R. Overman, et al. 1978. Retention of viruses during sludge application to soils. Prog. Water Technol. 10: 597-606.

Bitton, G., S.R. Farrah, R.H. Ruskin, et al. 1983. Survival of pathogenic and indicator organisms in groundwater. Ground Water. 21: 405-410.

Blacklow, N.R., G. Cukor, M.K. Bedigian, et al. 1979. Immune response and prevalence of antibody to Norwalk enteric virus as determined by radio-immunoassay. J. Clin. Microbiol. 10: 903-907.

Blake, P.A., D.T. Allegra, J.D. Synder, et al. 1980. Cholera -- A possible endemic focus in the United States. N. Engl. J. Med. 302: 305-309.

Blaser, M.J., F.M. Wilson and N.A. Wang. 1980. Reservoirs for human campylobacteriosis. J. Infect. Dis. 141: 665-669.

Boeck, W.C. 1921. On the longevity of human intestinal protozoan cysts. Am. J. Hyg. 1: 527-540.

Bottone, E.J. 1981. Yersinia enterocolitica. CRC Press, Boca Raton, FL.

Bouma, J., F.G. Baker and P.L.M. Veneman. 1974. Measurement of Water Movement in Soil Pedons. Univ. of Wisconsin-Extension, Madison, WI. p. 1-22.

Bouwer, H. 1984. Elements of soil science and groundwater hydrology. In: Groundwater Pollution Microbiology, G. Bitton and C.P. Gerba, Ed. John Wiley and Sons, New York. p. 9-38.

Boyd, J.W., T. Yoshida, L.E. Vereen, et al. 1969. Bacterial response to the soil environment. Sanitary Engr. Paper No. 5. Colorado State Univ., Fort Collins, CO.

CDC (Centers for Disease Control). 1979. Campylobacter enteritis in a household -- Colorado. CDC, Atlanta, GA. Morb. Mort. Wkly Rep. 28: 273-274.

CDC (Centers for Disease Control). 1985. Hepatitis Surveillance Report No. 49. CDC, Atlanta, GA.

Chang, S.L. and G.M. Fair. 1941. Viability and destruction of the cysts of Entamoeba histolytica. J. Am. Water Works Assoc. 33: 1705-1715.

Cherry, J.D. 1981. Nonpolio enteroviruses: Coxsackieviruses, echoviruses and enteroviruses. In: Textbook of Pediatric Infectious Diseases, R.D. Feigin and J.D. Cherry, Ed. W.B. Saunders, Philadelphia, PA. p. 1316-1365.

Chute, C., R. Smith and J. Baron. 1985. Risk factors for endemic giardiasis. Am. J. Epidemiol. 122: 515.

Colwell, R.R. 1984. Vibrios in the Environment. John Wiley and Sons, New York.

Cram, E.B. 1943. The effect of various treatment processes on the survival of helminth ova and protozoan cysts in sewage. Sewage Works J. 15: 1119-1138.

Crane, S.R. and J.A. Moore. 1984. Bacterial pollution of groundwater: A review. Water Air Soil Pollut. 22: 67-83.

Craun, G.F. 1984. Waterborne outbreaks of giardiasis: Current status. In: Giardia and Giardiasis, S.L. Erlandsen and E.A. Meyer, Ed. Plenum Press, New York. p. 243-261.

Craun, G.F. 1986. Waterborne Disease in the United States. CRC Press, Boca Raton, FL.

CST (Committee on Science and Technology). 1983. A review of risk assessment methodologies. U.S. House of Representatives, Washington, DC.

Damgaard-Larson, S., K.O. Hensen, E. Lund, et al. 1977. Survival and movement of enterovirus in connection with land disposal of sludges. Water Res. 11: 503-508.

D'Antonio, R.G., R.E. Winn, J.P. Taylor, et al. 1985. A waterborne outbreak of cryptosporidiasis in normal hosts. Ann. Internal Med. 103: 886-888.

D'Aoust, J.Y. 1985. Infective dose of Salmonella typhimurium in cheddar cheese. Am. J. Epidemiol. 122: 717-720.

Dazzo, F., P. Smith and D. Hubbell. 1972. Vertical dispersal of fecal coliforms in Scranton fine sand. Soil Crop Sci. Soc. Fl. Proc. 32: 99-102.

Diesch, S.L. 1971. Survival of leptospires in cattle manure. J. Am. Vet. Med. Assoc. 159: 1513-1517.

DOC-NOAA (Department of Commerce, National Oceanic and Atmospheric Administration). 1982. Comparative Climatic Data for the United States. DOC-NOAA, Asheville, NC.

Dominowska, C. and R. Malottke. 1971. Survival of Yersinia in water samples originating from various sources. Bull. Inst. Mar. Med. Gdansk. 22: 173-182.

Donnelly, J.A. and P.V. Scarpino. 1984. Isolation, characterization, and identification of microorganisms from laboratory and full-scale landfills. U.S. EPA, Cincinnati, OH. EPA-600/52-84-119.

Duboise, S.M., B.E. Moore, C.A. Sorber, et al. 1979. Viruses in soil systems. CRC Crit. Rev. Microbiol. 7: 245-285.

DuPont, H.L. and R.B. Hornick. 1973. Clinical approach to infectious diarrheas. Medicine. 52: 265-270.

Epstein, E. 1973. The physical processes in the soil as related to sewage sludge application. In: Recycling Municipal Sludges and Effluents on Land. National Assoc. of State Universities and Land-Grant Colleges, Washington, DC.

Erlandsen, S.L. and E.A. Meyer. 1984. Giardia and Giardiasis. Plenum Press, New York.

Evans, A.S. 1982. Epidemiological concepts and methods. In: Viral Infections of Humans, A.S. Evans, Ed. Plenum Press, New York. p. 3-13.

Farrah, S.R. and S.A. Schaub. 1983. Viruses in wastewater sludges. In: Viral Pollution of the Environment, G. Berg, Ed. CRC Press, Boca Raton, FL. p. 147-161.

Farrah, S.R., G. Bitton, E.M. Hoffman, et al. 1981. Survival of enteroviruses and coliform bacteria in a sludge lagoon. Appl. Environ. Microbiol. 41: 459-465.

Feachem, R.G., D.J. Bradley, H. Garelick, et al. 1983. Sanitation and disease. In: Health Aspects of Excreta and Wastewater Management. John Wiley and Sons, New York.

Findlay, C.R. 1973. Salmonellae in sewage sludge. II. Multiplication. Vet. Rec. 93: 102-103.

Furomoto, W.A. and R. Mickey. 1967. A mathematical model for the infectivity-dilution curve of tobacco mosaic virus in theoretical considerations. Virology. 32: 216-223.

Gary, G.W., J.C. Hierholzer and R.E. Black. 1979. Characteristics of non-cultivable adenoviruses associated with diarrhea in infants: A new subgroup of human adenovirus. J. Clin. Microbiol. 10: 96-103.

Gerba, C.P. 1983. Pathogens. In: Utilization of Municipal Wastewater and Sludge on Land, A.L. Page, et al., Ed. Univ. of California, Riverside, CA. p. 147-187.

Gerba, C.P. 1984a. Strategies for the control of viruses in drinking water. Report to the American Assoc. for the Advancement of Science, Environmental Science and Engineering Program, Washington, DC.

Gerba, C.P. 1984b. Applied and theoretical aspects of virus adsorption to surfaces. Adv. Appl. Microbiol. 30: 133-167.

Gerba, C.P. 1986a. Unpublished results. Univ. of Arizona, Tucson, AZ.

Gerba, C.P. 1986b. Transport and fate of viruses in soils: Field studies. In: Human Viruses in Sediments, Sludges and Soils, C. Rao, Ed. CRC Press, Boca Raton, FL. (In press)

Gerba, C.P. and G. Bitton. 1984. Microbial pollutants: Their survival and transport pattern to groundwater. In: Groundwater Pollution Microbiology, G. Bitton and C.P. Gerba, Ed. John Wiley and Sons, New York. p. 65-88.

Gerba, C.P., C. Wallis and J.L. Melnick. 1975. Fate of wastewater bacteria and viruses in soil. J. Irrig. Drain Div., ASCE. 101: 157-174.

Gerba, C.P., S.M. Goyal, C.J. Hurst, et al. 1979. Type and strain dependence of enterovirus adsorption to activated sludge, soils, and estuarine sediments. Water Res. 14: 1197-1198.

Gerba, C.P., S.M. Goyal, I. Cech, et al. 1982. Quantitative assessment of the adsorptive behavior of viruses to soils. Environ. Sci. Technol. 15: 940-944.

Gerba, C.P., S.N. Singh and J.B. Rose. 1985. Waterborne viral gastroenteritis and hepatitis. CRC Crit. Rev. Environ. Control. 15: 213-236.

Gerichter, C.B., I. Sechter, I. Gavish, et al. 1975. Viability of Vibrio cholerae biotype El Tor and of cholera phage on vegetables. Isr. J. Med. Sci. 11: 889-895.

Ghirose, W. 1986. Cornell Univ., Ithaca, NY. Personal communication.

Gilbert, R.G., C.P. Gerba, R.C. Rice, et al. 1976. Virus and bacteria removal from wastewater by land treatment. Appl. Environ. Microbiol. 32: 333-338.

Goyal, S.M. and C.P. Gerba. 1979. Comparative adsorption of human enteroviruses, simian rotavirus, and selected bacteriophages to soils. Appl. Environ. Microbiol. 38: 241-247.

Goyal, S.M., B.H. Keswick and C.P. Gerba. 1984. Viruses in groundwater beneath sewage irrigated cropland. Water Res. 18: 299-302.

Griffiths, H.J. 1978. A Handbook of Veterinary Parasitology of Domestic Animals of North America. Univ. of Minnesota, Minneapolis, MN.

Grohmann, G.S. 1985. The Norwalk and Norwalk-like viruses as etiologic agents in sporadic acute gastroenteritis. Presented at 185th Ann. Meeting of the Am. Soc. Microbiol., Las Vegas, NV. March.

Grondin, G. and C.P. Gerba. 1986. Virus dispersion in a coarse porous medium. Hydrology and water resources in Arizona and the southwest. J. Environ. Health. 16: 11-15.

Grosser, P.W. 1985. A one-dimensional mathematical model of virus transport. In: Proc. 2nd Int. Conf. Groundwater Quality Research, N.N. Durham and A.E. Redelfs, Ed. University Center for Water Research, Oklahoma State Univ., Stillwater, OK. p. 105-107.

Grunnet, K. and S.E. Olesen. 1979. Disappearance of microorganisms by infiltration and percolation of sewage. Prog. Water Technol. 4/5: 473-481.

Haas, C.N. 1983a. Estimation of risk due to low doses of microorganisms: A comparison of alternative methodologies. Am. J. Epidemiol. 118: 573-582.

Haas, C.N. 1983b. Effect of effluent disinfection on risks of viral disease transmission via recreational water exposure. J. Water Pollut. Control Fed. 55: 1111-1116.

Hagedorn, C. 1980. Potential health hazards associated with the disposal of sewage sludge on agricultural soils in western Oregon. Office of Water Research and Technology, Washington, DC. NTIS PB 80-220387.

Hammerberg, B., G.A. MacInnis and T. Hyler. 1978. Taenia saginata cysticerci in grazing steers in Virginia. J. Am. Vet. Med. Assoc. 173: 1462-1464.

Hazard, R.L. and M.D. Sobsey. 1985. Reduction of infectious hepatitis virus, poliovirus, and echovirus in miniature soil columns. In: Proc. Ann. Conf. American Water Works Assoc., Denver, CO.

Hendricks, D.W., F.J. Post and D.R. Khairnar. 1979. Adsorption of bacteria on soils. Water Air Soil Pollut. 12: 219-232.

Hess, E. and C. Breer. 1975. Epidemiology of salmonellae and fertilizing of grassland with sewage sludge. Zentralbl. Bakteriol. Parasitenk. D. Infektionskr. Hyg. 161: 54-60.

Highsmith, A.K., J.C. Feeley, P. Skaliy, et al. 1977. Isolation of Yersinia enterocolitica in well water and growth in distilled water. Appl. Environ. Microbiol. 34: 745-750.

Hudson, J.A. and H. Fennel. 1980. Disposal of sewage sludge to land: Chemical and microbiological aspects of sludge to land policy. J. Water Pollut. Control. 52: 370-387.

Hurst, C.J., C.P. Gerba and I. Cech. 1980a. Effects of environmental variables and soil characteristics on virus survival in soil. Appl. Environ. Microbiol. 40: 1067-1079.

Hurst, C.J., C.P. Gerba, J.C. Lance, et al. 1980b. Survival of enteroviruses in rapid-infiltration basins during the land application of wastewater. Appl. Environ. Microbiol. 40: 192-200.

Jackson, G.T., J.W. Bier and R.A. Rude. 1977. Recycling of refuse into the food chain: The parasite problem. In: Risk Assessment and Health Effects of Land Application of Municipal Wastewater and Sludges, B.P. Sagik and C.A. Sorber, Ed. Center for Applied Research and Technology, Univ. of Texas, San Antonio, TX.

Jakubowski, W. and J.C. Hoff. 1979. Waterborne Transmission of Giardiasis. U.S. EPA, Cincinnati, OH. EPA-600/9-79-001.

Jorgensen, P.H. 1985. Examination of the penetration of enteric viruses in soils under simulated conditions in the laboratory. Water Sci. Technol. 17: 197-199.

Jorgensen, P.H. and E. Lund. 1985. Detection and stability of enteric viruses in sludge, soil and groundwater. Water Sci. Technol. 17: 185-195.

Karaseva, E.V., Y.G. Chernukha and L.A. Piskunova. 1973. Results of studying the time of survival of pathogenic leptospira under natural conditions. J. Hyg. Epidemiol. Microbiol. Immunol. 17: 339-345.

Katz, M. and S.A. Plotkin. 1967. Minimal infective dose of attenuated poliovirus for man. Am. J. Public Health. 57: 1837-1840.

Keswick, B.H. and C.P. Gerba. 1980. Viruses in groundwater. Environ. Sci. Technol. 14: 1290-1297.

Keswick, B.H., C.P. Gerba, S.L. Secor, et al. 1982a. Survival of enteric viruses and indicator bacteria in groundwater. J. Environ. Sci. Health. A17: 903-912.

Keswick, B.H., D. Wang and C.P. Gerba. 1982b. The use of microorganisms as ground-water tracers: A review. Ground Water. 20: 142-149.

Kibbey, H.J., C. Hagedorn and E.L. McCoy. 1978. Use of fecal streptococci as indicators of pollution in soil. Appl. Environ. Microbiol. 35: 711-717.

Klein, D.A. and L.E. Casida, Jr. 1967. Escherichia coli dieout from normal soil as related to nutrient availability and the indigenous microflora. Can. J. Microbiol. 13: 1461-1470.

Koprowski, H., T.W. Norton, G.A. Jervis, et al. 1956. Clinical investigations of attenuated strains of poliomyelitis virus: Use as a method of immunization of children with living virus. J. Am. Med. Assoc. 160: 954-966.

Kowal, N.E. 1985. Health effects of land application of municipal sludge. U.S. EPA, Research Triangle Park, NC. EPA 600/1-85-015.

Lance, J.C. and C.P. Gerba. 1984. Virus movement in soil during saturated and unsaturated flow. Appl. Environ. Microbiol. 47: 335-337.

Lance, J.C., C.P. Gerba and J.L. Melnick. 1976. Virus movement in soil columns flooded with secondary sewage effluent. Appl. Environ. Microbiol. 32: 520-526.

Landry, E.F., J.M. Vaughn and W.F. Penello. 1980. Poliovirus retention in 75-cm soil cores after sewage and rainwater application. Appl. Environ. Microbiol. 40: 1032-1038.

Larkin, E.P., J.T. Tierney and R. Sullivan. 1976. Persistence of virus on sewage-irrigated vegetables. Proc. Am. Soc. Civil Engr. J. Environ. Engr. Div. 102: 29-35.

Lednar, W.M., S.M. Lemon, J.W. Kirkpatrick, et al. 1985. Frequency of illness associated with epidemic hepatitis A virus infections in adults. Am. J. Epidemiol. 122: 226-233.

Leong, L.Y.C. 1983. Removal and inactivation of viruses by treatment processes for potable water and wastewater: A review. Water Sci. Technol. 15: 91-114.

Lepow, M.L., R.J. Warren, V.G. Ingram, et al. 1962. Sabin type 1 (LSc2ab) oral poliomyelitis vaccine: Effect of dose upon response of newborne infants. Am. J. Dis. Child. 104: 67-71.

Lewis, W.J., S.S.D. Foster and B.S. Drasar. 1980. The risk of groundwater pollution by on-site sanitation in developing countries. IRCWD-Report No. 01/82. International Reference Centre for Wastes Disposal, Duebendorf, Switzerland.

Liew, P. and C.P. Gerba. 1980. Thermostabilization of enteroviruses by estuarine sediment. Appl. Environ. Microbiol. 40: 305-308.

Liu, D. 1982. The effect of sewage sludge land disposal on the microbiological quality of groundwater. Water Res. 16: 957-961.

Loehnert, E.P. 1981. Hohe nitratgehalte in einem landlichen gebiet in Nigeria verursacht durch ungeordnete ablagerung havslicher abfalle und exkremente. In: Quality of Groundwater, W. van Duijvenbooden, P. Glasbergen and H. van Lelyveld, Ed. Elsevier, Amsterdam. p. 147-153. (Ger.)

Logsdon, G.S., F.B. DeWalle and D.W. Hendricks. 1984. Filtration as a barrier to passage of cysts in drinking water. In: Giardia and Giardiasis, S.L. Erlandsen and E.A. Meyer, Ed. Plenum Press, New York. p. 287-309.

Mallman, W. and W. Litsky. 1951. Survival of selected enteric organisms in various types of soil. Am. J. Public Health. 41: 38-44.

Martin, G.N. and M.J. Noonan. 1977. Effects of domestic wastewater disposal by land irrigation on groundwater quality of the Central Canterbury Plains. Water Soil Tech. Publ. No. 7. National Water and Soil Conservation Organization, Wellington, New Zealand.

Matthess, G. and A. Pekdeger. 1985. Survival and transport of pathogenic bacteria and viruses in groundwater. In: Ground Water Quality, C.H. Ward, W. Giger and P.L. McCarty, Ed. John Wiley and Sons, New York. p. 472-482.

McCoy, E.L. and C.H. Hagedorn. 1980. Transport of resistance labeled Escherichia coli strains through a transition between two soils in a topographic sequence. J. Environ. Qual. 9: 686-691.

McFeters, G.A and D.G. Stuart. 1972. Survival of coliform bacteria in natural waters. Field and laboratory studies with membrane filter chambers. Appl. Microbiol. 24: 805-811.

McFeters, G.A., G.K. Bissonette, J.J. Jezeski, et al. 1974. Comparative survival of indicator bacteria and enteric pathogens in well water. Appl. Microbiol. 27: 823-829.

Metro (Municipality of Metropolitan Seattle). 1983. Health effects of sludge land application: A risk assessment. Municipality of Metropolitan Seattle Water Quality Division, Seattle, WA.

Meyer, R.B., K.D. Miller and E.S. Kaneshiro. 1978. Recovery of Ascaris eggs from sludge. J. Parasitol. 64: 380-383.

Minor, T.E., C.I. Allen, A.A. Tsiatis, et al. 1981. Human infective dose determination for oral poliovirus type 1 vaccine in infants. J. Clin. Microbiol. 13: 388-389.

Moore, B.E., B.P. Sagik and C.A. Sorber. 1977. Land application of sludges: Minimizing the impact of viruses on water resources. In: Risk Assessment and Health Effects of Land Application of Municipal Wastewater and Sludges, B.P. Sagik and C.A. Sorber, Ed. Center for Applied Research and Technology, Univ. of Texas, San Antonio, TX. p. 154-167.

Morris, R. and W.M. Waite. 1980. Evaluation of procedures for recovery of viruses from water. 2: Detection systems. Water Res. 14: 795-798.

Morris, J.G., Jr., R. Wilson, B.R. Davis, et al. 1981. Non-O group 1 Vibrio cholerae gastroenteritis in the United States. Ann. Intern. Med. 94: 656-658.

MSDGC (Metropolitan Sanitary District of Greater Chicago). 1979. Viral and bacterial levels resulting from land application of digested sludge. U.S. EPA, Cincinnati, OH. EPA 600/-79-015.

Murphy, W.H. and J.T. Syverton. 1958. Absorption and translocation of mammalian viruses by plants. II. Recovery and distribution of viruses in plants. Virology. 6: 623-636.

Musial, C.E. 1985. Development of a method for the detection of Cryptosporidium in water and selected studies on hepatitis A virus. Ph.D. Dissertation, Univ. of Arizona, Tucson, AZ.

Newton, W.L., H.J. Bennett and W.B. Figgat. 1949. Observations on the effects of various sewage treatment processes upon the eggs of Taenia saginata. Am. J. Hyg. 49: 166-175.

NRC (National Research Council). 1983. Risk Assessment in the Federal Government: Managing the Process. National Academy Press, Washington, DC.

NRC (National Research Council). 1985. An Evaluation of the Role of Microbiological Criteria for Foods and Food Ingredients. National Academy Press, Washington, DC.

Pahren, H. and W. Jakubowski. 1980. Wastewater Aerosols and Disease. U.S. EPA, Cincinnati, OH. EPA 600/9-80-028.

Peterson, D.A., L.G. Wolfe, E.P. Larkin, et al. 1978. Thermal treatment of hepatitis A virus in human feces. J. Med. Virol. 2: 201-206.

Rahe, T.M., C. Hagedorn, E.L. McCoy, et al. 1978. Transport of antibiotic-resistant Escherichia coli through western Oregon hillslope soils under conditions of saturated flow. J. Environ. Qual. 7: 487-494.

Reddy, K.R., R. Khaleel and M.R. Overcash. 1981. Behavior and transport of microbial pathogens and indicator organisms in soils treated with organic wastes. J. Environ. Qual. 10: 255-266.

Reimers, R.S., D.D. Bowman, A.J. Englande, et al. 1980. In: Sludge -- Health Risks of Land Application, G. Bitton, B.L. Damron, G.T. Edds and J.M. Davidson, Ed. Ann Arbor Science Publ., Ann Arbor, MI. p. 356.

Reimers, R.S., M.D. Little, A.J. Englande, et al. 1981. Parasites in Southern sludges and disinfection by standard sludge treatment. U.S. EPA, Cincinnati, OH. EPA 600/52-81-166.

Rudolfs, W., L.L. Falk and R.A. Ragotzkie. 1951a. Contamination of vegetables grown in polluted soil. III. Field studies on Ascaris eggs. Sewage Indust. Wastes. 23: 656-660.

Rudolfs, W.L., L.L. Falk and R.A. Ragotzkie. 1951b. Contamination of vegetables grown in polluted soil. II. Field and laboratory studies on Entamoeba cysts. Sewage Indust. Wastes. 23: 478-485.

Sagik, B.P., S.M. Duboise and C.A. Sorber. 1980. Health risks associated with microbial agents in municipal sludge. In: Sludge -- Health Risks of Land Application, G. Bitton, B.L. Damron, G.T. Edds and J.M. Davidson, Ed. Ann Arbor Science Publ., Ann Arbor, MI. p. 15-45.

Sandhu, S.S., W.J. Warren and P. Nelson. 1979. Magnitude of pollution indicator organisms in rural potable water. Appl. Environ. Microbiol. 37: 744-749.

Schiff, G.M., G.M. Stefanovic, E.C. Young, et al. 1984. Studies of echovirus-12 in volunteers: Determination of minimal infectious dose and the effect of previous infection on infectious dose. J. Infect. Dis. 150: 858-866.

Sedita, S.J., P. O'Brien, J.J. Bertucci, et al. 1977. Public health aspects of digested sludge utilization. In: Land as a Management Alternative, R.C. Loehr, Ed. Ann Arbor Science Publ., Ann Arbor, MI. p. 391-410.

Shablovskaya, E.A. 1963. Some data on the vertical migration of Strongyloides stercoralis in the soil. Tr. Ukr. Republikanskogo Nauchrogo Obshchestva. Parazitologov. 2: 74-80.

Siegel, G. 1982. Structure and biology of hepatitis A virus. In: Viral Hepatitis, W. Szmuness, H.J. Alter and J.E. Maynard, Ed. The Franklin Institute Press, Philadelphia, PA. p. 13-30.

Slonka, G.F., J.I. Moulthrop and L.W. Dewhirst. 1975. An epizootic of bovine cysticercosis. J. Am. Vet. Med. Assoc. 166: 678-681.

Smith, G.S., H.E. Kiosling, E.E. Ray, D.M. Hallford and C.H. Herbel. 1980. Fate of parasites in drying bed studies. In: Sludge -- Health Risks of Land Application, G. Bitton, B.L. Damron, G.T. Edds and J.M. Davidson, Ed. Ann Arbor Science Publ., Ann Arbor, MI.

Sobsey, M.D. 1983. Transport and fate of viruses in soils. In: Microbial Health Considerations of Soil Disposal of Domestic Wastewaters, L.W. Canter, et al., Ed. U.S. EPA, Cincinnati, OH. EPA 600/9-83-017. p. 175-197.

Sobsey, M.D. 1985. Univ. of North Carolina, Chapel Hill, NC. Unpublished results to James McNabb, U.S. EPA, Ada, OK, February 15 and August 15.

Sorber, C.A., B.E. Moore, D.E. Johnson, et al. 1984. Microbiological aerosols from the application of liquid sludge to land. J. Water Pollut. Control Fed. 56: 830-836.

Spinu, I., V. Topciu, T. Quy, et al. 1963. Man as a reservoir of infection in an epidemic of leptospirosis occurring in a jungle. Arch. Roum. Pathol. Exp. Microbiol. 22: 1081-1100.

Sterling, C. 1986. Tucson, AZ. Personal communication to C.P. Gerba, Univ. of Arizona, Tucson, AZ, January 15.

Storm, D.R., W.E. Sadler and D.K. Strathman. 1979. Helminth and index bacteria detection and enumeration. Sewage Sludge Management Program, Vol. 6. Sacramento Area Consultants, Sacramento, CA.

Stramer, S.L. 1984. Fates of poliovirus and enteric indicator bacteria during treatment in a septic tank system including septage disinfection. Ph.D. Dissertation, Univ. of Wisconsin-Madison, Madison, WI.

Tate, R.L. 1978. Cultural and environmental factors affecting the longevity of Escherichia coli in histosols. Appl. Environ. Microbiol. 35: 925-929.

Temple, K.L., A.K. Camper and G.A. McFeters. 1980. Survival of two enterobacteria in feces buried in soil under field conditions. Appl. Environ. Microbiol. 40: 794-797.

Theis, J.H., V. Bolton and D.R. Storm. 1978. Helminth ova in soil and sludge from twelve U.S. urban areas. J. Water Pollut. Control Fed. 50: 2485-2493.

U.S. EPA. 1974. Process Design Manual for Sludge Treatment and Disposal. Office of Technology Transfer, Cincinnati, OH. EPA 625/1-74-006.

U.S. EPA. 1978. Process Design Manual for Municipal Sludge Landfills. Office of Solid Waste, Cincinnati, OH. EPA 625/1-78-010.

Vaughn, J.M. and E.F. Landry. 1983. Viruses in soils and groundwater. In: Viral Pollution of the Environment, G. Berg, Ed. CRC Press, Boca Raton, FL. p. 163-210.

Vaughn, J.M., E.F. Landry, L.J. Baranosky, et al. 1978. Survey of human virus occurrence in wastewater-recharged groundwater on Long Island. Appl. Environ. Microbiol. 36: 47-51.

Vilker, V.L. 1981. Simulating virus movement in soils. In: Modeling Wastewater Renovation, I.K. Iskandar, Ed. John Wiley and Sons, New York. p. 223-253.

Vilker, V.L. and W.D. Burge. 1980. Adsorption mass transfer model for virus transport in soils. Water Res. 14: 783-790.

Wang, D.S., C.P. Gerba and J.C. Lance. 1981. Effect of soil permeability of virus removal through soil columns. Appl. Environ. Microbiol. 42: 83-88.

Wang, D.S., C.P. Gerba, J.C. Lance, et al. 1985. Comparative removal of enteric bacteria and poliovirus by sandy soils. J. Environ. Sci. Health. A20: 617-624.

Ward, R.L. and E.W. Akin. 1984. Minimum infectious dose of animal viruses. CRC Crit. Rev. Environ. Control. 14: 297-310.

Ward, R.L. and R.J. Mahler. 1982. Uptake of bacteriophage f2 through plant roots. Appl. Environ. Microbiol. 43: 1098-1103.

Ward, R.L., G.A. McFeters and J.G. Yeager. 1984. Pathogens in sludge: Occurrence, inactivation and potential for regrowth. Sandia Report SAND83-0557. TTC-0428. UC-71. Sandia National Laboratories, Albuquerque, NM.

Warren, K.S. 1974. Helminthic diseases endemic in the United States. Am. J. Trop. Med. Hyg. 23: 723-730.

Watson, D. 1980. The survival of salmonellae in sewage sludge applied to arable land. J. Water Pollut. Control Fed. 79: 11-18.

Wellings, F.M., A.L. Lewis and C.W. Mountain. 1974. Virus removal following wastewater spray irrigation of sandy soils. In: Virus Survival in Water and Wastewater Systems, J.F. Malina and B.P. Sagik, Ed. Univ. of Texas, Austin, TX. p. 253-260.

WHO (World Health Organization). 1981. The Risk to Health of Microbes in Sewage Sludge Applied to Land. Euro Reports and Studies 54. WHO, Copenhagen.

Williams, F.P. and E.W. Akin. 1986. Waterborne viral gastroenteritis. J. Am. Water Works Assoc. 78: 34-39.

Wright, W.H., E.B. Cram and M.O. Nolan. 1942. Preliminary observations on the effect of sewage treatment processes on the ova and cysts of intestinal parasites. Sewage Works J. 14: 1274-1280.

Yates, M.V. 1985. Septic tank siting to minimize the contamination of groundwater by microorganisms. American Assoc. for the Advancement of Science, Washington, DC.

Yates, M.V. and C.P. Gerba. 1985. Virus persistence in groundwater. Appl. Environ. Microbiol. 49: 778-781.

Yates, M.V., S.R. Yates, A.W. Warwick, et al. 1985. Preventing viral contamination of drinking water. In: Groundwater Contamination and Reclamation, K.C. Schmidt, Ed. American Water Resources Assoc., Bethesda, MD. p. 117-121.

Yates, M.V., S.R. Yates and J. Wagner. 1987. Modeling virus survival and transport in the subsurface. J. Contam. Hydrol. 1: 329-345.

Zerda, K.S., C.P. Gerba, K.C. Hov, et al. 1985. Adsorption of viruses to charge-modified silica. Appl. Environ. Microbiol. 49: 91-95.

Ziebell, W.A., D.H. Nero, J.F. Deininger, et al. 1975. Use of bacteria in assessing waste treatment and soil disposal systems. In: Home Sewage Disposal. American Society of Agricultural Engineers, Publ. 175, St. Joseph, MI. p. 58-63.

☆U.S. GOVERNMENT PRINTING OFFICE: 1988-548-150/87011

Agency Cincinnati OH 45268

BULK RATE
POSTAGE & FEES PAID
EPA
PERMIT No. G-35

Official Business
Penalty for Private Use, $300

Please make all necessary changes on the above label, detach or copy, and return to the address in the upper left-hand corner.

If you do not wish to receive these reports CHECK HERE ☐; detach, or copy this cover, and return to the address in the upper left-hand corner.

EPA/600/6-88/006